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Phenol

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Canadian Environmental Protection Act

PRIORITY SUBSTANCES LIST ASSESSMENT REPORT

Phenol

Environment Canada
Health Canada

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List of Acronyms and Abbreviations

B[a]P	benzo[a]pyrene
BCF	bioconcentration factor
CAS	Chemical Abstracts Service
CEPA	*Canadian Environmental Protection Act*
CFC	chlorofluorocarbon
CTV	Critical Toxicity Value
DMBA	dimethylbenz[a]anthracene
EEV	Estimated Exposure Value
ENEV	Estimated No-Effects Value
GWP	Global Warming Potential
ISCST3	Industrial Source Complex Short Term model
K_{oc}	organic carbon/water partition coefficient
K_{ow}	octanol/water partition coefficient
kg-bw	kilogram body weight
LC_{50}	median lethal concentration
LD_{50}	median lethal dose
LOAEL	Lowest-Observed-Adverse-Effect Level
LOEL	Lowest-Observed-Effect Level
MWTP	municipal wastewater treatment plant
NOAEL	No-Observed-Adverse-Effect Level
NOEL	No-Observed-Effect Level
ODP	Ozone Depletion Potential
POCP	Photochemical Ozone Creation Potential
PSL	Priority Substances List
TC	Tolerable Concentration
TI	Tolerable Intake
WCEM	Wildlife Contaminant Exposure Model

SYNOPSIS

Phenol is an aromatic alcohol with the chemical formula C_6H_6O. Although phenol is no longer produced in Canada, 76 000 tonnes were imported in 1995 and 95 000 tonnes in 1996. Manufacture of phenolic resins accounts for about 85% of phenol consumption.

Phenol is released to the Canadian environment as a by-product and contaminant from various industry sectors and from municipal wastewater treatment plants. The major industry sectors include the pulp, paper and wood products sector, the mineral (non-metallic) products sector, the chemical products sector, the steel and metal products sector, and the petroleum refining and products sector. In 1996, 321.8 tonnes of phenol/total phenolics were emitted to air and 58.5 tonnes released to water.

The environmental assessment was focussed on releases of phenol to air and water because the largest amounts of phenol are released to these media. Environmental effects are likely to occur close to release areas, because phenol has a short half-life in both air and water. Final effluent (i.e., end-of-pipe) concentrations of phenol from various industry sectors were used to estimate exposure of aquatic biota, because ambient water concentrations were not available. Exposure of terrestrial organisms was investigated for the highest emitters of phenol to air.

For aquatic organisms, the most sensitive assessment endpoint identified was mortality in embryos and larvae of rainbow trout. The meadow vole was selected as the herbivore most likely to be exposed to releases of phenol to air near point sources. Two exposure scenarios were investigated: 1) direct air inhalation, and 2) ingestion and direct air inhalation. The most sensitive organism exposed to phenol in soil is lettuce.

Results of the aquatic assessment demonstrated that for 22 out of 26 pulp and paper mills in Ontario, 6 out of 8 steel mill outfalls in Canada, and 14 out of 16 petroleum refining and production plants in Canada, the probability of phenol causing effects to greater than 5% of aquatic communities is negligible. Of the remaining, the probability of effects greater than 35% for early life stages of the most sensitive aquatic species exposed to phenol near outfalls was low (<5%). Effects near outfalls of municipal wastewater treatment plants due to phenol exposure are not likely.

Results of both exposure scenarios for herbivores demonstrated that phenol released by the highest emitters of phenol to air in Canada is unlikely to cause effects on terrestrial wildlife. Similarly, it is unlikely that phenol causes effects on terrestrial vegetation near high emitters.

Because of the reactivity of phenol in the atmosphere, its Photochemical Ozone Creation Potential is substantial. However, the quantities available for reaction make the contribution insignificant relative to those of other smog-forming substances. Reaction with ozone is negligible, and the absence of chlorine or bromine atoms in the molecule and the overall short half-life of phenol mean that its contributions to stratospheric ozone depletion and climate change are both negligible.

Available data upon which to base estimates of population exposure to phenol in Canada are limited; however, food appears to be the major route of exposure for members of the general population. Intakes are estimated to be elevated for populations in the vicinity of industrial point sources of phenol in Canada. Based on the results of studies conducted in experimental animals, the kidney appears to be a target organ for phenol-induced toxicity. Other

sensitive effects observed in laboratory mammals include histopathological changes in the liver and thymus, reduced counts of certain blood cells, suppressed immune response and effects on the nervous system. The estimated average daily intake by the general population from environmental sources and upper-bound estimates of exposures via inhalation for populations in the vicinity of industrial point sources are less than a Tolerable Intake derived on the basis of effect levels for non-neoplastic renal effects. A Tolerable Intake is the level of intake to which it is believed a person may be exposed daily over a lifetime without deleterious effect.

Based on information available, it is concluded that phenol is not entering the environment in a quantity or concentration or under conditions having or that may have an immediate or long-term harmful effect on the environment; constituting or that may constitute a danger to the environment on which human life depends; or constituting or that may constitute a danger in Canada to human life or health. Therefore, phenol is not considered to be "toxic" as defined in Section 11 of the *Canadian Environmental Protection Act* (CEPA).

The evaluation of options under CEPA to reduce exposure is not considered to be a priority at this time. However, this is based upon current use patterns; future releases of this compound should continue to be monitored to ensure that exposure does not increase to any significant extent.

1.0 INTRODUCTION

The *Canadian Environmental Protection Act* (CEPA) requires the federal Ministers of the Environment and of Health to prepare and publish a Priority Substances List (PSL) that identifies substances, including chemicals, groups of chemicals, effluents and wastes, that may be harmful to the environment or constitute a danger to human health. The Act also requires both Ministers to assess these substances and determine whether they are "toxic" as defined in Section 11 of the Act, which states:

> ...a substance is toxic if it is entering or may enter the environment in a quantity or concentration or under conditions
> (a) having or that may have an immediate or long-term harmful effect on the environment;
> (b) constituting or that may constitute a danger to the environment on which human life depends; or
> (c) constituting or that may constitute a danger in Canada to human life or health.

Substances that are assessed as "toxic" as defined in Section 11 may be placed on Schedule I of the Act and considered for possible risk management measures, such as regulations, guidelines or codes of practice to control any aspect of their life cycle, from the research and development stage through manufacture, use, storage, transport and ultimate disposal.

Based on initial screening of readily accessible information, the rationale for assessing phenol provided by the Ministers' Expert Advisory Panel on the Second Priority Substances List (Ministers' Expert Advisory Panel, 1995) was as follows:

> The organic chemical sector represents the largest source of this substance. Phenol is used primarily in the production of phenolic resins. It is also emitted from petroleum refineries, iron and steel mills, sewage treatment plants, and the dye and plastics industries. Acute and chronic ecotoxicological effects have been reported in fish, amphibians and reptiles exposed to phenol. In mammals, phenol affects the respiratory and nervous systems, liver and kidneys. An assessment is necessary to characterize exposure levels and risks to the environment and human health in Canada.

Descriptions of the approaches to assessment of the effects of Priority Substances on the environment and human health are available in published companion documents. A document entitled "Environmental Assessments of Priority Substances under the *Canadian Environmental Protection Act*. Guidance Manual Version 1.0 — March 1997" (Environment Canada, 1997a) provides guidance for conducting environmental assessments of Priority Substances in Canada. This document may be purchased from:

> Environmental Protection Publications
> Environmental Technology Advancement Directorate
> Environment Canada
> Ottawa, Ontario
> K1A 0H3

It is also available on the Internet at www.ec.gc.ca/cceb1/eng/psap.htm under the heading "Technical Guidance Manual."

The approach to assessment of effects on human health is outlined in the following publication of the Environmental Health Directorate of Health Canada: "*Canadian Environmental Protection Act* — Human Health Risk Assessment for Priority Substances" (Health Canada, 1994), copies of which are available from:

> Environmental Health Centre
> Room 104
> Health Canada
> Tunney's Pasture
> Ottawa, Ontario
> K1A 0L2

or on the Environmental Health Directorate publications web site (www.hc-sc.gc.ca/ehp/ehd/catalogue/bch.htm). The approach is also described in an article published in the *Journal of Environmental Science and Health — Environmental Carcinogenesis & Ecotoxicology Reviews* (Meek *et al.*, 1994). It should be noted that the approach outlined therein has evolved to incorporate recent developments in risk assessment methodology, which are described on the Environmental Substances Division web site (www.hc-sc.gc.ca/ehp/ehd/bch/env_contaminants/psap/psap.htm) and which will be addressed in future releases of the approach paper for the assessment of effects on human health.

The search strategies employed in the identification of data relevant to assessment of potential effects on the environment (prior to May 1998) and human health (prior to September 1997) are presented in Appendix A. Review articles were consulted where appropriate. However, all original studies that form the basis for determining whether phenol is "toxic" under CEPA have been critically evaluated by staff of Environment Canada (entry and environmental exposure and effects) and Health Canada (human exposure and effects on human health).

The environmental sections of this Assessment Report were produced by R. Breton, Environment Canada. A first draft of the exposure and effects chapters of the supporting documentation for the environmental assessment was prepared by Scott Teed and Dwayne Moore of The Cadmus Group, Inc.

An Environmental Resource Group was established by Environment Canada to assist in the writing, collection of exposure data and review of the environmental assessment for phenol. The Environmental Resource Group, which consisted of scientific experts from the federal government and industry, was established in the fall of 1996. Members included:

Roger Breton, Environment Canada
Lorna Brownlee, Environment Canada
Douglas Bryant, CanTox Inc.
Howard Carter, Imperial Oil
Jacques Gagnon, Natural Resources Canada
Patrick Georges, Consultant on behalf of Environment Canada
Roger Keefe, Imperial Oil
Tanis Lugsdin, Environment Canada
Sondra O'Block, Aristech Chemical Corporation
Lynne Patenaude, Environment Canada
Joe Wittwer, Environment Canada

Sections of this Assessment Report and the supporting documentation (Environment Canada, 1998a) relevant to the environmental assessment were reviewed by members of the Environmental Resource Group, as well as by:

Richard P. Brown, Exxon Biomedical Sciences, Inc.
Nigel Bunce, University of Guelph
John Headley, Environment Canada
Robert Kent, Environment Canada
Shawn Michajluk, Environment Canada
Dwayne Moore, The Cadmus Group, Inc.
Charles A. Staples, Assessment Technologies, Inc.
William Strachan, Environment Canada

The health-related sections of this Assessment Report and the supporting documentation were prepared by the following staff of Health Canada:

W. Bruce
M.E. Meek
R. Newhook

Sections of the Assessment Report and supporting documentation on genotoxicity were reviewed by D. Blakey of the Environmental and Occupational Toxicology Division of Health Canada. Sections of the supporting documentation pertaining to human health were reviewed externally by B. Duncan (Allied Signal, Inc.),

R. Gingell (Shell Chemical Company), G. Granville (Shell Canada Limited) and C. Morris (ICC U.S.A., Inc.), primarily to address adequacy of coverage. Accuracy of reporting, adequacy of coverage and defensibility of conclusions with respect to hazard characterization and dose–response analyses were considered in written review by staff of the Information Department of BIBRA International and at a panel meeting of the following members, convened by Toxicology Excellence in Risk Assessment (TERA):

M. Bogdanffy, DuPont Haskell Laboratory
M. Dourson, TERA
A. Jarabek, U.S. Environmental Protection Agency (written comments)
R. Keenan, ChemRisk Division of McLaren/Hart
G. Leikauf, University of Cincinnati
R. Manning, Georgia Department of Natural Resources
E. Ohanian, U.S. Environmental Protection Agency
K. Poirier, Procter and Gamble
A. Renwick, University of Southampton
L. Rosato, Millennium Petrochemical Corporation
L. Sirinek, Ohio Environmental Protection Agency

The health-related sections of the Assessment Report were reviewed and approved by the Health Protection Branch Risk Management meeting of Health Canada.

The entire Assessment Report was reviewed and approved by the Environment Canada/Health Canada CEPA Management Committee.

A draft of the Assessment Report was made available for a 60-day public comment period (May 1 to June 29, 1999) (Environment Canada and Health Canada, 1999). Following consideration of comments received, the Assessment Report was revised as appropriate. A summary of the comments and their responses is available on the Internet at:

www.ec.gc.ca/cceb1/eng/final/index_e.html

The text of the Assessment Report has been structured to address environmental effects initially (relevant to determination of "toxic" under Paragraphs 11(a) and (b)), followed by effects on human health (relevant to determination of "toxic" under Paragraph 11(c)).

Copies of this Assessment Report are available upon request from:

Inquiry Centre
Environment Canada
Main Floor, Place Vincent Massey
351 St. Joseph Blvd.
Hull, Quebec
K1A 0H3

or on the Internet at:

www.ec.gc.ca/cceb1/eng/final/index_e.html

Unpublished supporting documentation, which presents additional information, is available upon request from:

Commercial Chemicals Evaluation Branch
Environment Canada
14th Floor, Place Vincent Massey
351 St. Joseph Blvd.
Hull, Quebec
K1A 0H3

or

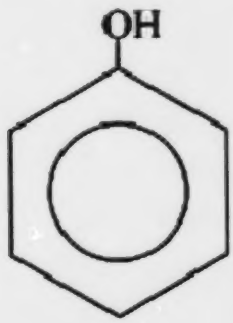
Environmental Health Centre
Room 104
Health Canada
Tunney's Pasture
Ottawa, Ontario
K1A 0L2

2.0 Summary of Information Critical to Assessment of "Toxic" under CEPA

2.1 Identity and physical/chemical properties

The chemical formula of phenol is C_6H_6O, and its molecular weight is 94.11. Its Chemical Abstracts Service (CAS) number is 108-95-2. The chemical structure of phenol is presented in Figure 1. Common names for phenol include benzene phenol, benzenol carbolic acid, hydrobenzene, monohydroxybenzene, monophenol, phenic acid, phenol alcohol, phenyl hydroxide and phenylic acid (Environment Canada, 1998a).

Figure 1 Chemical structure of phenol



Phenol is a white to light pink crystalline solid and has a characteristic acrid smell and a sharp burning taste. The physical and chemical properties of phenol are presented in Table 1. Additional information on the physical and chemical properties of phenol can be found in the following reviews: Shiu *et al.* (1994), Mackay *et al.* (1995), DMER and AEL (1996) and Environment Canada (1998a).

Since the focus of this assessment is the single substance phenol, every effort was made to distinguish phenol from total phenolics. Total phenolics are defined as mixtures of substituted aromatic alcohols. Many data reported concerning environmental levels and releases do not make that distinction. Confusion may arise for many reasons, one of which is that the analytical method used to obtain data is often not reported. When phenol data were not available, total phenolics data were used as surrogates.

2.2 Entry characterization

2.2.1 Production, importation and uses

Phenol is no longer produced in Canada. The last two plants that manufactured phenol closed in 1992 (SRI International, 1993). As reported in a survey of Canadian industry carried out under authority of Section 16 of CEPA, phenol was imported mostly in its pure form but also as various types of resins and polymers in amounts greater than 76 000 tonnes in 1995 and 95 000 tonnes in 1996. About 98% of phenolic substances were imported in the pure form of phenol (Environment Canada, 1997b).

Phenol is a common industrial chemical. In Canada, the manufacture of phenolic resins accounts for about 85% of phenol consumption (Environment Canada, 1997b). Most of the phenolic resins produced are used for oriented strand board production, panels, insulation, and articles and formulations such as paints, lubricants, creams, adhesives, brakes, electrical components and electrodes (SRI International, 1993). The largest amounts of phenolic resins are produced in Ontario (108 000 tonnes), Quebec (105 000 tonnes), British Columbia (35 000 tonnes) and Alberta (18 000 tonnes) (Camford Information Services, 1994; SRI International, 1994).

Phenol is also used in a wide range of other applications, including as a feedstock in the production of other organic substances (including

TABLE 1 Physical and chemical properties of phenol[1]

Property	Value	Reference
Melting point (°C)	41	Verschueren, 1983
Boiling point (°C)	182	Verschueren, 1983
Vapour pressure (Pa)	47	Dean, 1985
Henry's law constant (Pa·m³/mol)	0.059	Abd-El-Bary *et al.*, 1986
pK_a	9.99	Dean, 1985
Log K_{oc}	1.15–3.49	DMER and AEL, 1996
Log K_{ow}	1.46	Fujita *et al.*, 1964
Solubility in water (mg/L)	88 360	Blackman *et al.*, 1955

[1] See supporting documentation from Environment Canada (Environment Canada, 1998a) for a more complete listing of ranges of values reported and criteria for selection of physical and chemical properties.

bisphenol A, caprolactam, aniline, adipic acid, alkyl phenols, chlorophenols and other chemicals) and in the production or manufacture of adhesives, explosives, coke, fertilizers, illuminating gas, paints and paint removers, rubber, asbestos goods, wood preservatives, textiles, drugs, pharmaceutical preparations, perfumes and bakelite (Deichmann and Keplinger, 1981; Environment Canada, 1998a). Phenol is also used as a general disinfectant, anesthetic and antiseptic and is present in a number of consumer products, including ointments, ear and nose drops, cold sore lotions, mouthwashes, gargles, toothache drops, analgesic rubs, throat lozenges and sprays, and antiseptic lotions (Gosselin *et al.*, 1984; Reynolds, 1989; Gennaro, 1990).

Phenol may be used as a formulant in pesticides and can be formed during the degradation of certain pesticides in soil. The assessment of the environmental risks posed by phenol as a component of pesticides is the responsibility of the Pest Management Regulatory Agency under the *Pest Control Products Act*. Therefore, an assessment of the potential health and environmental effects from the pesticidal use of phenol was not carried out under CEPA.

2.2.2 *Sources and releases*

2.2.2.1 Natural sources

Phenol may occur naturally in water and soil as the decomposition product of plants, vegetation and animal waste (Dobbins *et al.*, 1987). It is released to the environment by these natural sources, mostly in trace amounts of phenolic substances (CCREM, 1987). Increased natural environmental concentrations may also result from forest fires (IPCS, 1994a).

2.2.2.2 Anthropogenic sources

Phenol is produced as an intermediate in the preparation of other chemicals and can be released as a by-product or contaminant. In 1996, total releases of 414.7 tonnes of phenol/total phenolics were reported to Environment Canada through surveys of industry carried out for this assessment under the authority of Section 16 of CEPA (Table 2).

Based on the information presented in Table 2 and on complementary information from the National Pollutant Release Inventory and the Accelerated Reduction/Elimination of Toxics program (Environment Canada, 1998a), the environmental assessment focussed on releases of phenol to air and water, because the largest amounts of phenol are released to these media.

TABLE 2 1996 releases of phenol/total phenolics by industry sector[1]

Industry sector	Releases (tonnes)			
	Air	Water	Other[2]	Total releases
Pulp, paper and wood	205.6	44.3	6.5	256.4
Mineral (non-metallic)	62.7	0.0	1.3	64.0
Chemical	22.9	0.0	15.4	38.3
Steel and metal	23.3	9.2	2.0	34.5
Petroleum refining	1.8	5.0	6.5	13.3
Other[3]	5.5	0.0	2.7	8.2
Total	**321.8**	**58.5**	**34.4**	**414.7**

[1] From survey of Canadian industry carried out under Section 16 of CEPA (Environment Canada, 1997b).

[2] Includes discharges of industrial effluents to municipal wastewater treatment plants (MWTPs), disposal to landfills, deep well injection and land for land farming.

[3] Represents the textile products sector, the transportation equipment sector, the industrial machinery and equipment sector and the instruments and related products sector.

Although not shown in Table 2, phenol is also released from municipal wastewater treatment plants (MWTPs). In 1995, the amount of total phenolics released from four MWTPs in British Columbia was estimated to be between 14 and 16 tonnes (GVRD, 1996). Information on loadings of phenol/total phenolics for MWTPs across Canada is not available. Concentrations of phenol/total phenolics in final effluent are presented in Section 2.3.2.4.

2.3 Exposure characterization

2.3.1 Environmental fate

Several reviews of the scientific literature on degradation rates of phenol in various environmental compartments have been prepared (Shiu *et al.*, 1994; Mackay *et al.*, 1995; DMER and AEL, 1996). Photooxidation in air and biodegradation in water and soil are expected to be the major removal processes. The fate of phenol in each environmental compartment is summarized briefly below.

2.3.1.1 Air

In the atmosphere, phenol exists predominantly in the vapour phase (Eisenreich *et al.*, 1981). The estimated half-life in air generally varies depending upon specific atmospheric conditions (e.g., temperature, time of year), and values ranging from 2.28 to 22.8 hours for reaction with hydroxyl radicals have been reported in the literature (RIVM, 1986; Howard, 1989; Howard *et al.*, 1991). DMER and AEL (1996) and Shui *et al.* (1994) suggested a half-life of 17 hours for phenol in air for photooxidation. Because of its short half-life, phenol is not expected to be transported over great distances in the atmosphere. Phenol has the potential to be removed from the atmosphere via photooxidation by reaction with hydroxyl and nitrate radicals, photolysis, and wet and dry deposition (Atkinson *et al.*, 1987, 1992; Bunce, 1996; Van Dusen, 1996).

2.3.1.2 Water

Phenol reacts as a weak acid in water but, based upon its high pK_a (9.99), is not expected to dissociate in the pH range typical in the natural

environment. Phenol may undergo numerous removal processes, including biodegradation, photooxidation, photolysis and volatilization.

Biodegradation is a major process for the removal of phenol from surface waters (Hwang *et al.*, 1986; U.S. EPA, 1990), provided the concentration is not high enough to cause significant inhibition (ATSDR, 1989). Ananyeva *et al.* (1992) observed that phenol is non-persistent in water and will completely biodegrade over a period of approximately 70 hours. The suggested half-life of phenol in water is 55 hours (Mackay *et al.*, 1995; DMER and AEL, 1996). Phenol generally reacts with hydroxyl and peroxyl radicals and singlet oxygen in sunlit surface waters (Scully and Hoigné, 1987; IPCS, 1994a). Half-lives were reported to be 100 hours for the hydroxyl radical reaction and 19.2 hours for the peroxyl radical reaction (Howard, 1989). In estuarine waters, photolysis is a minor transformation process, particularly in summer when biodegradation predominates (Hwang *et al.*, 1986). The half-life of phenol due to volatilization from surface water is 3.2 months (U.S. EPA, 1990).

The estimated half-life of phenol in groundwater ranges from 12 to 168 hours (Howard *et al.*, 1991). Many factors influence the potential for groundwater contamination, including soil depth and type and microbial abundance (Aelion *et al.*, 1987; Dobbins *et al.*, 1987; IPCS, 1994a).

2.3.1.3 Sediments

Phenol has a low organic carbon/water partition coefficient (log K_{oc} = 1.15–3.49) and a low octanol/water partition coefficient (log K_{ow} = 1.46) and is therefore not expected to significantly adsorb to suspended or bottom sediment in water (U.S. EPA, 1990; DMER and AEL, 1996). DMER and AEL (1996) and Shiu *et al.* (1994) suggested a biodegradation half-life of 550 hours for phenol in sediment.

2.3.1.4 Soils

The physical and chemical properties of soil that influence the fate and behaviour of phenol are pH, exchange capacity, organic matter, clay content and texture. Phenol may undergo numerous removal processes, including biodegradation, adsorption/desorption, volatilization and oxidation. These processes determine the mobility (e.g., leaching), distribution and persistence of phenol in soil.

Phenol is an abundant naturally occurring chemical that tends to biodegrade rapidly in the environment (Baker and Mayfield, 1980; Dobbins *et al.*, 1987; Howard, 1989). Both aerobic and anaerobic soil microorganisms are capable of utilizing phenol as a growth substrate, although decomposition is more rapid under aerobic conditions than under anaerobic conditions (Scott *et al.*, 1982; Howard, 1989). The reported biodegradation half-life of phenol in various soil types ranges from 2.7 to 552 hours (Alexander and Aleem, 1961; Scott *et al.*, 1982; Federle, 1988; Howard *et al.*, 1991; Loehr and Mathews, 1992; DMER and AEL, 1996). The suggested half-life of phenol in soil is 170 hours (Mackay *et al.*, 1995; DMER and AEL, 1996). Degradation of phenol decreases with increasing concentration, indicating inhibition of biodegradation by phenol itself (Scott *et al.*, 1982; Dean-Ross, 1989).

Artiola-Fortuny and Fuller (1982) evaluated the adsorptive capabilities of five soils for phenol and found that the two most important soil properties controlling adsorption by mineral soils are percent iron oxides and pH. Phenol was adsorbed more strongly to soil under high pH (ionized form) and high iron oxide content. Adsorption reduces the rate of biodegradation in soils, although sorption to clay surfaces is reversible (Saltzman and Yariv, 1975; Knezovich *et al.*, 1988). Because phenol has a low log K_{oc} and a low log K_{ow}, sorption to organic matter is expected to be low. In acidic soil, low adsorption and high mobility have been reported for phenol (Scott *et al.*, 1982; Howard, 1989).

Because of a moderate vapour pressure (47 Pa) and a Henry's law constant of 0.059 Pa·m³/mol, volatilization of phenol is expected to be rapid from dry near-surface soils (IPCS, 1994a). Phenol is sensitive to oxidizing agents (including metal ions, such as manganese and iron) and may auto-oxidize to form coloured complexes such as quinones (Rineheart, 1973; IPCS, 1994a).

2.3.1.5 Biota

Log bioconcentration factors (BCFs) are reported to range from 0.88 to 5.09 in the scientific literature for a variety of organisms (Environment Canada, 1998a). The higher values in this range were derived from ^{14}C concentration measurements that include both phenol plus metabolites. As a result, BCF values derived from such studies may overestimate the accumulation of phenol. The most appropriate range of log BCFs for phenol is therefore 0.88–2.44. Based on the suggested log K_{ow} (1.46) (Fujita *et al.*, 1964), a BCF of 7.6 has been calculated (Veith *et al.*, 1980; Lyman *et al.*, 1990). Based on these data, phenol is not expected to bioaccumulate significantly (Verschueren, 1983; Budavari *et al.*, 1989; Howard, 1989; IPCS, 1994a; DMER and AEL, 1996).

2.3.1.6 Environmental partitioning

Fugacity modelling was carried out to provide an overview of key reaction, intercompartment and advection (movement out of a system) pathways for phenol and its overall distribution in the environment (DMER and AEL, 1996). A steady-state, non-equilibrium model (EQC Level III fugacity model) was run using the methods developed by Mackay (1991) and Mackay and Paterson (1991). Values for input parameters were as follows: molecular weight, 94.11 g/mol; water solubility, 88 360 mg/L; vapour pressure, 47 Pa; log K_{ow}, 1.46; Henry's law constant, 0.059 Pa·m³/mol; half-life in air, 17 hours; half-life in water, 55 hours; half-life in soil, 170 hours; half-life in sediment, 550 hours. Modelling was based on an assumed default emission rate of 1000 kg/hour into a region of 100 000 km², which includes a surface water area (20 m deep) of 10 000 km². The height of the atmosphere was assumed to be 1000 m. Sediments and soils were assumed to have an organic carbon content of 4% and 2% and a depth of 1 cm and 10 cm, respectively. The estimated percent distribution predicted by this model is not affected by the assumed emission rate.

Modelling indicates that phenol partitions differently depending on the medium to which it is released. For example, if emitted into air, about half of phenol is found in air, while most of the remainder is found in soil. When phenol is released to water, the model predicts that almost all of it is found in water (Mackay *et al.*, 1995; DMER and AEL, 1996). The results were expected, given the low Henry's law constant and the high water solubility of phenol. The assessment of releases to air, therefore, focussed on environmental exposure pathways involving contact with both air and soil, while the assessment of releases to water focussed on exposure to water only. Because of the short half-lives in these media, the highest exposures are likely to be in areas close to releases (DMER and AEL, 1996), and the environmental assessment was focussed on these areas.

If reliable data on discharge quantities are available, the environmental concentrations predicted by models for a given region of Canada can be compared with monitoring data that may exist for the region. This was done using the CHEMCAN4 Level III fugacity model, which includes in its assumptions the dimensions and environmental parameters for various contiguous regions of Canada. The region modelled was southern Ontario, the region of Canada for which total releases are the largest (Environment Canada, 1998a), and the only one for which monitoring data removed from point sources are available (Section 2.3.2). The chemical-specific properties and degradation rates were the same as those used with the EQC model described above. Based on the total estimated releases in this region to air (45.9 tonnes per year) and to water

(16.8 tonnes per year) from the 1993 National Pollutant Release Inventory, the Level III fugacity modelling predicted approximate phenol concentrations of 2×10^{-4} µg/m³ in ambient air, 1×10^{-4} µg/L in water, 4×10^{-6} µg/g in soil, 6×10^{-5} µg/g in terrestrial plants and 4×10^{-5} µg/g in terrestrial animals (DMER and AEL, 1996).

2.3.2 Environmental concentrations

2.3.2.1 Ambient air

Data concerning levels of phenol in ambient air are very limited but indicate that concentrations are low except in the vicinity of point sources. In a limited survey conducted in Windsor in 1992 at 10 urban and 2 rural locations, phenol was detected (although not accurately quantified) at approximate instantaneous concentrations ranging from not detected (detection limit approximately 1 µg/m³) to 3.1 µg/m³ (OMEE, 1994). The mean ambient atmospheric level of phenol was 0.12 µg/m³ (range not detected [detection limit not specified] to 0.18 µg/m³) in seven samples from one U.S. urban/suburban site (Columbus, Ohio) in 1974 (Jones, 1976). In a secondary account of monitoring at five urban sites in Santa Clara County, California, not known to be near point sources, phenol was not detected in any of a total of 22 samples (detection limit 0.25 µg/m³) (Hunt *et al.*, 1988).

Somewhat higher airborne concentrations of phenol have been measured in very limited short-term surveys in the vicinity of point sources in Canada and the United States. In air samples collected from three areas in Alberta, most of which were obtained in the vicinity of industrial plants, concentrations of phenol were 6.6 µg/m³ in two samples from the site of a wood treatment plant (reaching 476 µg/m³ in two samples adjacent to the train carrying the treated wood immediately after removal from the processing chamber), 4.3 µg/m³ in a sample from the main street in the nearby town and 16.1 µg/m³ in a single sample collected near a rape seed plant; phenol was not detected (detection limit not specified) in nine samples collected near two gas processing plants (Strosher, 1982). In surveys of ambient air near point sources in Ontario, 30-minute average concentrations of phenol downwind of two phenol-formaldehyde resin plants ranged from 7.3 to 36 µg/m³ (n = 4; De Brou and Bell, 1987) and from 4 to 57 µg/m³ (n = 18; De Brou and Ng, 1989). Levels were not detectable near one plant when it was not operating (detection limit approximately 1 µg/m³) (De Brou and Bell, 1987), while concentrations in 17 samples of ambient air near a wood treatment facility ranged from not detected (method detection limit 0.1–10 µg/m³) to 9 µg/m³ (De Brou, 1990). In U.S. surveys reported in a secondary source, concentrations of phenol averaged 106 µg/m³ in 83 samples from seven areas in the immediate vicinity of phenol manufacturing and/or processing plants monitored between 1974 and 1978 (Brodzinsky and Singh, 1983).

Owing to the paucity of suitable monitoring data collected near point sources in Canada, air dispersion models were used to estimate concentrations of phenol in ambient air near facilities that emitted the greatest quantities of phenol. Values for input parameters were obtained directly from the companies and from the responses to the Section 16 Notices under CEPA. Because of the confidentiality of some information on entry and exposure data used in the assessment, industrial sources are referred to as Company 1 for the highest emitter of phenol to air, Company 2 for the second highest, etc.

The SCREEN3 model (U.S. EPA, 1995) was used to provide worst-case one-hour ground-level concentrations nearest the three highest emitters. The worst-case concentrations were estimated by Environment Canada to be 228 µg/m³ (phenol) at a distance of 100 m from the stack for Company 1, 0.066 µg/m³ (phenol) at a distance of 1272 m from the stack for Company 2 and 0.065 µg/m³ (total phenolics) at a distance of 618 m from the stack for Company 3.[1] Because the

[1] To run the SCREEN3 model, input data were obtained directly from the company or through surveys of industry carried out under the authority of Section 16 of CEPA.

predicted concentration for Company 1 is much higher (four orders of magnitude) than those for Companies 2 and 3, the focus for the remainder of the modelling was Company 1.

The Industrial Source Complex Short Term (ISCST3) model was used to provide a more realistic estimate of the concentrations in air at various locations near Company 1 (Davis, 1997). The model was run for each hour of a 5.5-year period using meteorological data from the nearest station to the site and specific source information (i.e., stack and building parameters, emission parameters). The maximum predicted on-site 24-hour average concentration of phenol was 145 μg/m³ 112 m southwest of the stack, on the roof of the building. However, concentrations between 135 and 145 μg/m³ occurred at a frequency of only 0.1% of the time over the entire 5.5 years at this location, and the mean predicted concentration was 12 μg/m³. The predicted concentrations decreased substantially as distance from the stack increased, from a maximum 24-hour average of 22 μg/m³ in the nearest field outside the plant boundaries 0.75 km from the stack to around 0.5 μg/m³ at a distance of 8 km from the stack. The median concentration in the nearest field was 0.007 μg/m³. The distributions were right skewed (i.e., the majority of the samples contained low concentrations).

2.3.2.2 Indoor air

Identified data on concentrations of phenol in indoor air are limited, with no relevant studies conducted in Canada. In samples of vented air taken during two different classes in a college auditorium in Iowa, U.S.A., the average concentrations of phenol were 18 and 16 μg/m³, compared with 4 μg/m³ when the auditorium was vacant (Wang, 1975). Phenol was detected in indoor air samples collected in 1992–93 from an office building in Silver Spring, Maryland, U.S.A., in which employees had complaints of ill feeling, and phenol was a component of the epoxy floor-levelling material used on the floors (Martin *et al.*, 1994). Average concentrations of phenol in the air samples ranged from 12 to 78 μg/m³, with 1060 μg/m³ detected in one sample from an area in which the rug adhesive was being removed from the levelling material.

In a survey in Zagreb, Yugoslavia, concentrations of phenol were measured by a colorimetric technique over several days inside and outside of 6 office buildings and in 10 kindergartens and 8 schools (Kalinic *et al.*, 1987). The average concentration of phenol outside the office buildings was 51 μg/m³, compared with 7 μg/m³ indoors. In naturally ventilated buildings, there was a stronger correlation between indoor and outdoor concentrations of phenol than for buildings that were air conditioned, suggesting that a considerable portion of the indoor contamination may have been due to outside sources. In one of the office buildings, higher levels of phenol (mean 18 μg/m³) may have been due to recent construction (phenol-formaldehyde resins are used in the construction industry). In the kindergartens, the mean concentration of phenol was 18 μg/m³ in the summer and 6 μg/m³ in the winter, compared with mean levels in the schools of 4 μg/m³ in the summer and 7 μg/m³ in the winter.

2.3.2.3 Drinking water

Phenol has seldom been detected in Canadian drinking water supplies. In the most recent national survey, phenol was not detected (quantitation limit 0.004 μg/L) in 120 samples of treated water taken between October 1984 and July 1985 from 40 potable water treatment facilities across Canada (Sithole and Williams, 1986).

Phenol was only occasionally detected at low levels in regional Canadian surveys. Trace amounts were detected (detection limit 1.0 μg/L) in 67 of 2022 treated water samples taken from sites throughout Alberta between 1987 and 1994 (Alberta Environmental Protection, 1996). Levels of phenol in tap water and in finished water at treatment facilities throughout Quebec, surveyed between 1985 and 1994 (111 samples in total), ranged from <0.025 to 1.1 μg/L, with most containing <0.4 μg/L (Riopel, 1994). Phenol was not detected in three tap water samples from

TABLE 3 Range of highest concentrations per source in Canada

Source	Range of highest mean[1] concentrations across facilities (mg/L)	Chemical(s)	Years(s), number of facilities
Pulp, paper and wood products sector	ND[2]–0.40	Phenol	1996 data for 26 mills in Ontario
Steel and metal products sector	0.006–0.34	Total phenolics	1995–1997 data for 8 outfalls (4 mills) in Canada
Petroleum refining and products sector	0.0004–2.03[3]	Total phenolics	1993–1996 data for 16 refineries in Canada
MWTPs	0.002–2.60[4]	Total phenolics	1985–1997 data for 31 MWTPs in Canada

[1] For example, if an industry sector has five plants in Canada and 12 monthly mean concentrations are available per plant, then the highest concentration was selected for each plant; the range of concentrations would therefore represent five values.

[2] ND = not detected (detection limit 0.0024 mg/L).

[3] The percent phenol in total phenolics is estimated at 11% (OMOE, 1992a).

[4] The percent phenol in total phenolics is estimated at 1% (OMOE, 1988).

Toronto in 1988 (detection limit 0.000 04 µg/L) or in seven major brands of bottled spring water delivered to Toronto homes (detection limit 0.001 µg/L) (Kendall, 1990).

2.3.2.4 Surface water

Natural phenol/total phenolics concentrations in surface water across Canada are generally below 0.002 mg/L (CCREM, 1987). Between 1975 and 1995, concentrations of total phenolics in several thousand surface water samples from locations removed from point sources were generally below 0.001 mg/L (OMOE, 1991a,b,c,d; B.C. MOE, 1996; Environment Canada, 1998a). Because no recent data on ambient concentrations of phenol in water were available, final effluent concentrations (i.e., end-of-pipe) were used as a measure of exposure in surface waters near point sources. This section is focussed, therefore, on concentrations in final effluents.

2.3.2.4.1 Industrial and municipal wastewater treatment plant effluents

Recent final effluent concentrations have been collected for the industry sectors that discharge the most phenol/total phenolics to surface waters in Canada — i.e., the pulp, paper and wood products, the steel and metal products, and the petroleum refining and products sectors. The highest monthly average concentrations in final effluents for facilities in each sector are presented in Table 3. Also included in Table 3 is a range of concentrations (grab and 24-hour composites) for 31 MWTPs in Canada.

In Canada, there are over 150 pulp, paper and wood products mills (Southam Inc., 1997). Final effluent concentrations of phenol for 1996 are available for the 26 pulp, paper and wood products mills in Ontario (Table 3) (OMOE, 1997). These concentrations are expected to be representative of mills in Canada, since all mills have installed appropriate treatment systems as a result of the Pulp and Paper Effluent Regulations

under the *Fisheries Act* that came into force on December 31, 1995 (Environment Canada, 1996).

Phenol is released from integrated steel mills as a result of the cokemaking, ironmaking and sintering production operations (OMOE, 1991e; Bhargava, 1993; OMEE, 1995). The four integrated steel mills in Canada are located in Ontario. Some of these mills have more than one final effluent outfall. Total phenolics concentrations are presented for the eight outfalls for the four Ontario mills (Table 3) (OMOE, 1997).[2] Information on the percentage of phenol in total phenolics for this sector is not available.

Final effluent concentrations of total phenolics are available for the 16 petroleum refineries in Canada (Table 3) (Environment Canada, 1997c). During a 12-month study in 1989–90, the average concentration of phenol in final effluents measured in seven refineries in Ontario was 0.0012 mg/L. The average total phenolics concentration for the seven refineries was 0.0113 mg/L. Based on these averages, the percent phenol in total phenolics in effluent from refineries is approximately 11% (OMOE, 1992a).

Final effluent concentrations of total phenolics are available for 31 MWTPs located in Ontario, British Columbia and Alberta (OMOE, 1988; B.C. MOE, 1996; GVRD, 1996).[3] Total phenolics concentrations reported for these MWTPs ranged from 0.002 to 2.6 mg/L (Environment Canada, 1998a). In 1987, the highest measured final effluent concentrations of phenol for seven MWTPs in Ontario ranged from 0.0041 to 0.0173 mg/L. For five of those seven MWTPs, concentrations of phenol and total phenolics are available. Based on the average concentrations, the percent phenol in total phenolics in final effluents for the five MWTPs is 1% (OMOE, 1988). In 1997, nine 24-hour composite samples and grab samples in final effluents were taken at two MWTPs in Ontario. In all cases, phenol was not detected (detection limit 0.0017 mg/L) (Environment Canada, 1998b).

2.3.2.5 Groundwater

Phenol was present at levels ranging from not detected (detection limit not reported) to 72.6 µg/L in 11 groundwater samples from two abandoned sand and gravel pits near Ville-Mercier, Quebec, into which approximately 40 000 m^3 of liquid wastes, including phenolic compounds, were dumped between 1968 and 1972 (Pakdel *et al.*, 1992).

2.3.2.6 Sediments

Concentrations of phenol ranging from <0.0001 (detection limit) to 0.033 mg/kg dry weight in sediment from various rivers in Quebec were reported by Laliberté (1990). Phenol was also detected in the sediments of the Ottawa River basin at levels up to 0.2 mg/kg dry weight (Paul and Laliberté, 1987). In British Columbia, concentrations of phenol in sediment from the Fraser River estuary ranged from 0.007 to 0.056 mg/kg dry weight (Hall *et al.*, 1986). Levels of phenol ranged from 0.07 to 0.27 mg/kg dry weight in marine sediments near artificial island platforms for exploratory drilling in the Beaufort Sea (Fowler and Hope, 1984).

2.3.2.7 Soils

Information on levels of phenol in Canadian soils is limited. Phenol was not detected (detection limit 0.1 mg/kg) in 30 soil samples collected in 1987 in Port Credit, Oakville and Burlington, Ontario, from urban residential and parkland locations (Golder Associates, 1987). In 60 samples of soil collected from areas classified as old urban parkland (>40 years) in Ontario, the 98th percentile of concentrations was 0.027 mg/kg (OMOE, 1992b).

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[2] Data also provided by individual steel companies in 1997.

[3] Data also obtained by Environment Canada through 1997 surveys sent to municipalities having a population greater than 5000; additional data provided by a regional municipality of Ontario in 1998.

The concentration of phenol in 16 samples representing typical agricultural soils from eight provinces ranged from not detected in one sample (method detection limit 0.02 mg/kg dry weight) to 0.92 mg/kg dry weight. In intensively cropped soils from southern Ontario farms where there had been repeated heavy use of pesticides for many years, concentrations of phenol ranged from not detected (in three of the six samples) to 1.10 mg/kg dry weight (Webber, 1994; Webber and Wang, 1995).

Because of a lack of data near industrial point sources, the ISCST3 model was also used to predict where the maximum total deposition of phenol would occur near Company 1 (identified as the largest source of emissions to the Canadian atmosphere) over a 5.5-year period (Davis, 1997). Deposition estimates were also predicted for other locations, including the nearest field outside the plant boundaries. The predicted depositions of phenol decreased rapidly as the distance from the stack increased. The distributions at all locations were right skewed — i.e., few occurrences with high concentrations were predicted. These soil deposition estimates were used to help set up a soil sampling program undertaken for Environment Canada. In surface soil samples (5 cm depth), phenol was detected at two on-site locations at concentrations of 1.7 and 1.1 mg/kg dry weight and was not detected at 13 other on- and off-site locations (detection limit 0.1 mg/kg dry weight) (Géologos Inc., 1997).

From 1968 to 1972, approximately 40 000 m^3 of liquid wastes, among them phenolic compounds, were dumped in two abandoned sand and gravel pits near Ville-Mercier, Quebec (Pakdel *et al.*, 1992). Phenol was detected at concentrations ranging from 0.0011 to 0.0092 mg/kg in soil samples from 11 areas; however, levels as high as 12.4 mg/kg were measured at one extremely contaminated site.

2.3.2.8 Biota

Concentrations of phenol in aquatic organisms have been reported for only a few species in Canada. Whole-body concentrations ranged from 0.004 to 0.32 mg/kg wet weight in starry flounder (*Platichthys stellatus*) in the Fraser River estuary (Hall *et al.*, 1986). Similarly, in common carp (*Cyprinus carpio*) and channel catfish (*Ictalurus punctatus*) from the Grand River, Ontario, mean whole-body concentrations were 0.16 and 0.06 mg/kg wet weight, respectively (Camanzo *et al.*, 1987).

No empirical data are available for terrestrial biota. However, using maximum (44.5 mg phenol/m^2) and median (3.05 mg phenol/m^2) total deposition estimates generated by the ISCST3 model and estimates of biomass per unit area, steady-state concentrations of phenol in vegetation in the field nearest to Company 1 were estimated to be 148 and 1.3 mg/kg wet weight, respectively (Section 3.1.2.2).

2.3.2.9 Foods

In two Canadian surveys, samples of 33 food groups (each a composite of individual food items, combined in approximate proportion to their consumption in the Nutrition Canada Survey) were collected from retail outlets in Calgary, Alberta, in the spring of 1991 and in Windsor, Ontario, in January 1992 (ETL, 1991, 1992). Phenol was not detected in the majority of the composites in both surveys. In the more recent survey, in Windsor, less than 1 µg phenol/g was detected in organ meats, cured pork, canned weiners, cheese and butter, and alcoholic drinks, while less than 0.1 µg/g was detected in fat and oil products, sugar and jam products, and non-alcoholic drinks (the exact levels measured are reported in the footnote to Table 6).

2.3.2.10 Consumer products

Phenol is an effective disinfectant against vegetative Gram-positive and Gram-negative bacteria and some fungi and viruses (Reynolds, 1989). Weak solutions of phenol have been used topically for disinfection, and a 5% solution has been used as a disinfectant for excreta. Other disinfectants contain 20–50% phenol compounds,

and fungicides contain 2–4.5% phenol (Gosselin *et al.*, 1984). Phenol is also present in antibacterial preparations that are given topically or orally. In liquid formulations, it is present at about 2%, and in medicinal tablets and lozenges to 33 mg in each tablet/lozenge (Gosselin *et al.*, 1984). Phenol is commonly used as an antipruritic (relieves itching) and is used in the form of phenolated calamine lotion (1%), phenol ointment (2%) or a simple aqueous solution (0.5–1%) (Gennaro, 1990). Phenol is an active ingredient in a number of non-prescription drugs in Canada, including lip balms, ointments and throat sprays or lozenges (Kealey, 1997).

Phenol is also a constituent of tobacco smoke. Mainstream smoke contains 9–161 µg per cigarette and 35–110 µg per cigar, while 100–420 µg phenol per cigarette are released in sidestream smoke (ATSDR, 1989).

2.3.3 Endogenous production

In addition to exposure of humans to exogenous phenol in environmental media, phenols are produced in the metabolism of tyrosine by bacteria in the gut. This exposure is proportional to the amount of protein consumed. Mean excretion rates of phenol in healthy subjects ranged from 1 to 9.8 mg/day (Bone *et al.*, 1976; Lawrie and Renwick, 1987; Renwick *et al.*, 1988), with an upper range of 39 mg/day given in one study (Renwick *et al.*, 1988).

2.4 Effects characterization

2.4.1 Ecotoxicology

2.4.1.1 Aquatic organisms

The acute toxicity of phenol to freshwater organisms has been extensively studied. Fish have been sensitive to phenol at concentrations ranging from 5.02 mg/L (96-hour LC_{50}) for rainbow trout (*Oncorhynchus mykiss*) to 85 mg/L (2.5-hour LC_{50}) for goldfish (*Carassius auratus*) (McLeay, 1976; Kishino and Kobayashi, 1995). Toxicity to freshwater invertebrate species ranges from 2 mg/L (48-hour LC_{50}) for three caddisfly and one mayfly species to 2000 mg/L (48-hour LC_{50}) for the flower fly (*Eristalis* sp.) (Kamshilov and Flerov, 1978).

Freshwater algae are less sensitive than fish and invertebrates to acute exposures of phenol in fresh water. Effect concentrations of phenol range from 7.5 mg/L (for cell multiplication inhibition) for *Scenedesmus quadricauda* to 1211 mg/L (24-hour EC_{50}, assimilation) for *Scenedesmus subspicatus* (Bringmann and Kühn, 1980; Tisler and Zagorc-Koncan, 1995). Effect concentrations of phenol for aquatic macrophytes and vascular plants reported in the scientific literature range from 3 mg/L (12–14 days, abnormal growth) for *Lemna perpusilla* to 1500 mg/L (48-hour EC_{50}, chlorosis) for *Lemna minor* (Blackman *et al.*, 1955; Rowe *et al.*, 1982).

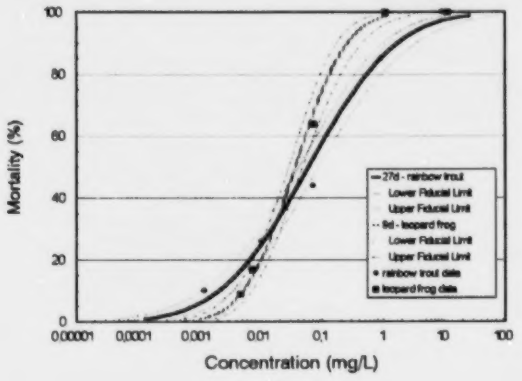
Based on the limited information available on the acute toxicity of phenol to marine organisms, the sensitivities of fish and invertebrates are similar (Environment Canada, 1998a). For example, after acute exposures, effect concentrations in marine fish ranged from 5.6 mg/L (96-hour LC_{50}) to 30.6 mg/L (96-hour LC_{50}) (Kondaiah and Murty, 1994). In marine invertebrates, adverse acute effects occurred at concentrations ranging from 32 mg/L (96-hour $LC_{5.6}$) to 172 mg/L (24-hour LC_{50}) (Key, 1981; Smith *et al.*, 1994). Effect concentrations for marine algae range from 7.8 mg/L (11-day Lowest-Observed-Effect Level, or LOEL) to 49.8 mg/L (5-day EC_{50}) (Thursby *et al.*, 1985; Cowgill *et al.*, 1989).

The chronic toxicity of phenol to freshwater organisms has also been reported in several studies. The embryo-larval stage of several species was particularly sensitive. Amphibians had 5- to 9-day LC_{50}s ranging from 0.04 to 11.23 mg/L measured under flow-through conditions (Birge *et al.*, 1980). The 6.5- to 58-day LC_{50}s in fish range from 0.07 to 2.67 mg/L measured under flow-through conditions (Birge *et al.*, 1979; DeGraeve *et al.*, 1980; Millemann *et al.*, 1984). Deneer *et al.* (1988) reported that

there was an adverse effect on growth (EC_{10}) of *Daphnia magna* exposed to as little as 0.46 mg phenol/L for 16 days. A 16-day No-Observed-Effect Level (NOEL) of 0.16 mg/L was reported.

The two most sensitive endpoints are the embryo-larval stages of leopard frog (*Rana pipiens*) and rainbow trout exposed to phenol until four days posthatch. For rainbow trout, the tests were conducted under hardwater (200 mg $CaCO_3$/L) and softwater (50 mg $CaCO_3$/L) conditions. Because of the controversy in the scientific literature regarding water hardness and how it affects the toxicity of phenol, only the toxicity results for which rainbow trout showed greater sensitivity are presented. In this case, rainbow trout was more sensitive under hardwater conditions. The 9-day LC_{50} for leopard frog was 0.04 mg/L, and rainbow trout had a 27-day LC_{50} of 0.07 mg/L (Birge *et al*., 1979, 1980). The data presented by Birge *et al.* (1979, 1980) for both species were analysed using PROC NLIN in SAS using log-probit and log-logistic analysis.[4] The results of this analysis are presented in Figure 2. Both models shown in the figure had adequate fit ($p > 0.05$). The 9-day LC_{25} for leopard frog was determined to be 0.0128 mg/L, while the 27-day LC_{25} for rainbow trout was 0.01 mg/L.

FIGURE 2 Comparison of phenol concentration–response distributions for 27-day exposure to embryo-larval stage of rainbow trout and 9-day exposure to embryo-larval stage of leopard frog



2.4.1.2 Terrestrial organisms

Effects on terrestrial organisms have been reported by several investigators. Increased mortality was observed in earthworms exposed to concentrations ranging from 188 to 6900 mg/kg (Neuhauser *et al.*, 1986; Neuhauser and Callahan, 1990; Environment Canada, 1995). The most sensitive earthworm species is *Eudrilus eugeniae*, with a 14-day LC_{50} of 188 mg/kg dry weight in artificial soil (Neuhauser *et al.*, 1986). Environment Canada (1995) reported a 14-day LC_{25} of 210 mg/kg dry weight in artificial soil for *Eisenia fetida*. Artificial soil tests conducted on terrestrial plants resulted in chronic effect concentrations ranging from 79 to 170 mg/kg dry weight (Environment Canada, 1995). Of the plant species examined, lettuce (*Lactuca sativa*) was the most sensitive, with a 5-day EC_{25} of 79 mg/kg dry weight for inhibition of seedling emergence (Environment Canada, 1995). Phenol inhibits nitrification significantly, especially at concentrations higher than 500 mg/kg (Beccari *et al.*, 1980; Den Blanken, 1993).

Only one study was identified in which the effects of phenol on wildlife were investigated. The 18-hour LD_{50} for the red-winged blackbird (*Agelaius phoeniceus*) was reported to be >113 mg/kg (Schafer *et al.*, 1983).

2.4.2 Abiotic atmospheric effects

Worst-case calculations were made to determine if phenol has the potential to contribute to depletion of stratospheric ozone, ground-level ozone formation or climate change (Bunce, 1996).

The Ozone Depletion Potential (ODP) is 0, since phenol is not a halogenated compound. It will therefore not contribute to the depletion of stratospheric ozone.

[4] Analysis conducted by The Cadmus Group, Inc., Ottawa, Ontario, 1997.

The Photochemical Ozone Creation Potential (POCP) was estimated to be 98 (relative to the value of an equal mass of the reference compound ethene, which has a POCP of 100), based on the following formula:

$$POCP = (k_{phenol}/k_{ethene}) \times (M_{ethene}/M_{phenol}) \times 100$$

where:

- k_{phenol} is the rate constant for the reaction of phenol with OH radicals (2.8×10^{-11} cm³/mol per second),
- k_{ethene} is the rate constant for the reaction of ethene with OH radicals (8.5×10^{-12} cm³/mol per second),
- M_{ethene} is the molecular weight of ethene (28.1 g/mol), and
- M_{phenol} is the molecular weight of phenol (94.1 g/mol).

The Global Warming Potential (GWP) was calculated to be 3.4×10^{-5} (relative to the reference compound CFC-11, which has a GWP of 1), based on the following formula:

$$GWP = (t_{phenol}/t_{CFC\text{-}11}) \times (M_{CFC\text{-}11}/M_{phenol}) \times (S_{phenol}/S_{CFC\text{-}11})$$

where:

- t_{phenol} is the lifetime of phenol (0.0014 years),
- $t_{CFC\text{-}11}$ is the lifetime of CFC-11 (60 years),
- $M_{CFC\text{-}11}$ is the molecular weight of CFC-11 (137.5 g/mol),
- M_{phenol} is the molecular weight of phenol (94.1 g/mol),
- S_{phenol} is the infrared absorption strength of phenol (2389/cm² per atmosphere, default), and
- $S_{CFC\text{-}11}$ is the infrared absorption strength of CFC-11 (2389/cm² per atmosphere).

Results indicate that, because of its reactivity in the atmosphere, phenol has a significant Photochemical Ozone Creation Potential of 98 (the value for an equal mass of ethene is 100); however, the quantities available for reaction make the contribution insignificant relative to those of other smog-forming substances. Because phenol has a very short atmospheric lifetime, a negligible Global Warming Potential (3.4×10^{-5}) was estimated (Bunce, 1996). Owing to phenol's short photooxidation half-life of 17 hours in the atmosphere (Shiu *et al.*, 1994; DMER and AEL, 1996) and to the absence of chlorine and bromine in the molecule, the Ozone Depletion Potential was estimated to be 0 (Bunce, 1996).

2.4.3 *Experimental mammals and* in vitro

2.4.3.1 Acute toxicity and irritation

Phenol is moderately acutely toxic in laboratory animals, with oral LD_{50} values for rats and rabbits falling within a narrow range, from 0.34 to 0.65 g/kg-bw (Deichmann and Witherup, 1944; Liao and Oehme, 1981; Berman *et al.*, 1995; Moser *et al.*, 1995). Owing to phenol's rapid absorption through skin, LD_{50} values for the dermal route are similar to those for oral exposure, ranging from 0.5 to 0.68 mL/kg-bw (Conning and Hayes, 1970; Brown *et al.*, 1975). Effects observed in rats following acute oral or dermal exposure included those on the nervous system and lesions in the liver, kidney, spleen and thymus. LC_{50} values after inhalation exposure to phenol were 316 and 177 mg/m³ for rats and mice, respectively (Nagornyi, 1976).

Following dermal or ocular exposure to phenol, local irritation has been observed in rabbits, rats, mice and pigs at concentrations as low as 1%, depending upon the vehicle used. Irritant effects on the skin include severe skin lesions, edema, erythema and necrosis (Deichmann, 1949; Deichmann *et al.*, 1950, 1952; Conning and Hayes, 1970; Flickinger, 1976; Pullin *et al.*, 1978; Patrick *et al.*, 1985).

Phenol has not induced sensitization in available studies in guinea pigs and mice (Itoh, 1982; Descotes, 1988; Dunn *et al.*, 1990).

2.4.3.2 Short-term and subchronic toxicity

The short-term toxicity of inhaled phenol was recently investigated in Fischer 344 rats exposed to phenol vapour, by nose only, 6 hours per day, 5 days per week, for 2 weeks, followed by a 14-day recovery period (CMA, 1998). No adverse effects on body weights, food consumption, clinical pathology, organ weights, gross pathology or histopathology in several tissues were observed at the highest dose tested, which was 98 mg/m^3 (No-Observed-Adverse-Effect Level, or NOAEL); no Lowest-Observed-Adverse-Effect Level (LOAEL) was reported (CMA, 1998).[5]

Other identified data on the toxicity of phenol following repeated exposure by inhalation are extremely limited, being confined to a small number of studies, most of them poorly documented, in which single dose levels were administered. Short-term or subchronic exposure to 100–200 mg/m^3 affected the nervous system and damaged the heart, lungs, liver and kidneys in guinea pigs and rabbits, but not in rats, in one study (Deichmann *et al.*, 1944) and caused hepatic damage and transient neurological impairment in rats in another study (Dalin and Kristoffersson, 1974).

The available database on repeated-dose toxicity following oral administration is limited to several early subchronic investigations or more recent short-term specialized investigations of neurological and immunological effects.

In small groups (n = 8) of female Fischer 344 rats exposed to 0, 4, 12, 40 or 120 mg phenol/kg-bw per day by gavage in a water vehicle for 14 days, there were no histological or neurobehavioural effects observed at 12 mg/kg-bw per day other than in one animal of the eight, with necrosis of the thymus. However, at 40 mg/kg-bw per day, pathological changes in the kidneys were observed (two animals with tubular degeneration in the papillar region, and one with protein casts in the tubules), along with thymic necrosis in two of eight animals, hepatic necrosis in one animal and (presumably hepatic) vacuolar degeneration in one animal. There were no pathological changes in control rats (Berman *et al.*, 1995). The nervous system was also affected by phenol in this study, as indicated by significant inhibition of pupil response at 4 days at the highest dose, alterations in the activity domain at 9 and/or 15 days at the two highest doses, and non-significantly decreased motor activity and increased rearing at day 15 in the rats receiving 40 mg/kg-bw per day (all rats from 120 mg/kg-bw per day dose group were dead by day 11) (Moser *et al.*, 1995). The NOAEL and LOEL are considered to be 12 mg/kg-bw per day and 40 mg/kg-bw per day, respectively.[6]

Suppression of immune response was observed in a study of groups of five male CD-1 mice continuously exposed to 0, 1.8, 6.2 or 33.6 mg phenol/kg-bw per day in drinking water for four weeks (Hsieh *et al.*, 1992). Although there were no overt clinical signs of toxicity, no effects on food or water consumption and no gross lesions or organ weight changes at autopsy in any treatment group, there was significant suppression of antibody production (IgM antibody plaque-forming cells per 10^6 splenocytes, and serum anti-sheep erythrocyte antibody) in response to a T-cell-dependent antigen (sheep red blood cells) at the two highest dose levels. A range of other immune effects was observed at the highest dose only, including significant suppression of the stimulation of cultured

[5] This study was not fully reported at the time of completion of the health risk assessment and was therefore not considered in the hazard evaluation and dose–response analyses for effects on human health.

[6] Because of the small group sizes, the NOAEL for histopathological changes in the various organs is unclear, but it might be considered as 12 mg/kg-bw per day. The reporting of the effect levels at which behavioural effects were observed in the original report is somewhat unclear, but the authors determined the "NOAEL" for these effects to be 12 mg/kg-bw per day (Moser *et al.*, 1995).

splenic lymphocytes by the B-cell mitogen lipopolysaccharide, the T-cell mitogen phytohemagglutinin and the T- and B-cell mitogen pokeweed, but not by concanavalin, and significant suppression of the proliferative ability of splenic lymphocytes in response to alloantigens (mixed lymphocyte response). Circulating erythrocyte counts decreased significantly compared with controls in all dose groups in a dose-dependent manner, whereas the hematocrit was significantly suppressed in the highest dose group only. Decreases in a number of measured neurochemicals were observed in the brain, restricted primarily to the highest dose group, although levels of dopamine in the corpus striatum were significantly decreased in all dose groups. The LOEL for immune suppression is considered to be 6.2 mg/kg-bw per day, with a LOEL for hematological and neurobiochemical effects of 1.8 mg/kg-bw per day. The observed immunological effects, which were considered to be of uncertain clinical significance, might be stress related. Moreover, there were seeming inconsistencies in the data; the decrease in circulating red blood cell counts at lower doses was not accompanied by associated decreases in hematocrit, a pattern often associated with compensatory increased cellularity due to decreased consumption of drinking water, although no effect on water consumption was observed in this study.

In an early subchronic study, in rats receiving 50 or 100 mg phenol/kg-bw by gavage in a water vehicle five days per week over six months, there were mild histopathological changes in the liver and mild to moderate kidney damage, accompanied by small increases in liver and kidney weight, at the highest dose (statistical significance not reported) (Adams, 1944). The LOEL for histopathological effects on the kidney is considered to be 50 mg/kg-bw per day.

In contrast, in a range-finding study by the National Toxicology Program (NCI, 1980), no gross or microscopic changes were noted, in an extensive array of tissues, in rats or mice of both sexes exposed for 13 weeks to phenol in drinking water at concentrations ranging from 100 to 10 000 mg/L (estimated to be approximately equivalent to doses of 14–819 mg/kg-bw per day in rats and 19–380 mg/kg-bw per day in mice; ATSDR, 1989). Decreased water intake and depressed body weight gain (in rats, 16% for males and 26% for females; in mice, 80% for males and 33% for females) were observed at the highest concentration (statistical significance not reported). The NOAEL (based on linear interpolation from the highest dose group) and LOAEL are considered to be approximately 236 mg/kg-bw per day and 819 mg/kg-bw per day, respectively, in rats and 124 mg/kg-bw per day and 380 mg/kg-bw per day, respectively, in mice, based on effects on body weight accompanied by decreased water consumption.

In a study limited to investigation of effects on the hematopoietic system, thrombocytopenia, mild eosinophilia, reticulocytosis and a decrease in the index of maturation of erythroblasts in the bone marrow were reported in guinea pigs administered 0.5 or 40 mg phenol/kg-bw per day orally for 3.5 months, although only the first effect was clearly dose related and reported as statistically significant (Sudakova and Nosova, 1981). The LOEL for hematopoietic effects is considered to be 0.5 mg/kg-bw per day.

2.4.3.3 Chronic toxicity/carcinogenicity

The chronic oral toxicity and carcinogenicity of phenol in laboratory animals were investigated in two early studies in Fischer 344 rats and $B6C3F_1$ mice, performed by the National Toxicology Program (NCI, 1980). After exposure of rats to phenol in drinking water for two years at concentrations of 2500 or 5000 mg/L (estimated intakes approximately equivalent to doses of 280 and 630 mg/kg-bw per day; ATSDR, 1989), a decrease in mean body weight was observed in both sexes from the high-dose group from week 20 onwards. Water consumption in the low- and high-dose groups was reduced; however, food consumption was comparable to controls. Significantly increased incidences of pheochromocytomas of the adrenal medulla, C-cell carcinomas of the thyroid, interstitial

cell tumours in the testes and leukemias or lymphomas were observed in the low-dose males only. There was a high spontaneous rate of leukemias or lymphomas observed in the matched controls of the study relative to historical controls. No non-neoplastic histopathological effects were observed in either dose group, and no dose-related effects on survival were noted. Based on the high spontaneous rate for leukemias or lymphomas observed in the control animals and the lack of a positive effect for the various tumour types in the high-dose group, the authors concluded that under the conditions of the bioassay, phenol was not carcinogenic for either male or female rats. The NOAEL and LO(A)EL for non-neoplastic effects (decreased body weight) in rats are considered to be 280 mg/kg-bw per day and 630 mg/kg-bw per day, respectively.

When mice were administered drinking water containing 2500 or 5000 mg phenol/L (estimated to be approximately equivalent to doses of 356 and 523 mg/kg-bw per day; ATSDR, 1989) for two years, no treatment-related increase in the incidence of tumours at any site was observed in either sex (NCI, 1980). No non-neoplastic histopathological effects were observed in either dose group. There were dose-related decreases in water consumption and in mean body weight gain in all groups, while food consumption in the two dose groups was comparable to controls. The LO(A)EL for non-neoplastic effects (decreased body weight gain, accompanied by decreased water consumption) in mice is considered to be 356 mg/kg-bw per day.

Phenol has been tested in a series of dermal initiation/promotion studies, in which different strains of mice were exposed to a single initiating dose of dimethylbenz[a]anthracene (DMBA) or benzo[a]pyrene (B[a]P), followed by repeated application of solutions of phenol for between 32 and 72 weeks (Salaman and Glendenning, 1957; Boutwell and Bosch, 1959; Wynder and Hoffmann, 1961). In these studies, phenol promoted the development of both malignant and benign DMBA- or B[a]P-induced tumours. In most of the strains studied, tumours were not observed in mice exposed to the phenol solution alone. When tumours occurred in the phenol-exposed mice, they were mostly benign and restricted almost entirely to concentrations of phenol (10–20%) that cause severe damage to the skin.

In two studies of the cocarcinogenesis of phenol in mice, a slight reduction in B[a]P-induced tumours was noted when phenol and B[a]P were applied simultaneously for 368 or 460 days, but phenol alone induced either no tumours or only a single papilloma (Van Duuren *et al.*, 1971, 1973; Van Duuren and Goldschmidt, 1976).

2.4.3.4 Genotoxicity

In non-mammalian test systems, predominantly negative results have been obtained for gene mutations in bacteria after exposure to phenol, including in the standard Ames test in *Salmonella typhimurium* in the presence or absence of metabolic activation from Aroclor-induced rat and hamster liver (IARC, 1989; IPCS, 1994a) and in tests for reverse mutations in *Escherichia coli* (JETOC, 1996).

Phenol has been genotoxic in numerous *in vitro* studies in mammalian cell lines. An increase in mutation frequency was observed in Chinese hamster fibroblasts (Paschin and Bahitova, 1982), mouse lymphoma cells (Wagenheim and Bolcsfoldi, 1988) and Syrian hamster embryo cells (Tsutsui *et al.*, 1997) after incubation with phenol with or without a metabolic activation system. Chromosomal aberrations, sister chromatid exchanges and/or micronucleus formation were induced in hamster ovary cells (Ivett *et al.*, 1989; Miller *et al.*, 1995), in Syrian hamster embryo cells (Tsutsui *et al.*, 1997) and in some studies in human lymphocytes (Morimoto and Wolff, 1980; Morimoto *et al.*, 1983, 1993; Erexson *et al.*, 1985; Jansson *et al.*, 1986; Yager *et al.*, 1990) after *in vitro* exposure to phenol. Phenol induced DNA damage or inhibited DNA synthesis in various mammalian cells *in vitro* (Painter and Howard, 1982; Pellack-Walker *et al.*, 1985; Garberg *et al.*, 1988; Reddy *et al.*, 1990; Kolachana *et al.*, 1993).

In *in vivo* studies in mice, small but statistically significant increases in micronuclei were evident at similar intraperitoneal doses in five studies from four authors (120 mg/kg-bw, Marrazzini *et al.*, 1994; 265 mg/kg-bw, Ciranni *et al.*, 1988a,b; 80 mg/kg-bw, Shelby *et al.*, 1993; and 160 mg/kg-bw, Chen and Eastmond, 1995). Results were negative in three additional investigations of suboptimal design, in which effects were examined at only one time point (Gocke *et al.*, 1981; Gad-El Karim *et al.*, 1986; Barale *et al.*, 1990).

In a multigeneration study in mice, daily doses as low as 2 mL of a 0.08 mg phenol/L solution (approximately equivalent to 6.4 µg/kg-bw per day) for five generations resulted in a dose-dependent increase in the frequency of chromosomal aberrations in spermatogonia and primary spermatocytes of male mice within each generation, along with an increase in aberration frequency in successive generations (Bulsiewicz, 1977). No other endpoints were measured in the study. In the only other *in vivo* study in which the occurrence of chromosomal aberrations after phenol exposure was investigated, no increases were observed in bone marrow of rats after oral or intraperitoneal exposure to as much as 510 and 180 mg phenol/kg-bw, respectively (Thompson and Gibson, 1984).

In other *in vivo* studies in mammals, there were no increases in DNA single strand breaks in rat testicular DNA (Skare and Schrotel, 1984) and in mouse bone marrow (Kolachana *et al.*, 1993) at intraperitoneal doses of 75–79 mg/kg-bw or in DNA adducts in bone marrow, Zymbal gland, liver or spleen after administration of phenol to female rats in four daily oral doses of 75 mg/kg-bw (Reddy *et al.*, 1990).

2.4.3.5 Reproductive and developmental toxicity

No histopathological effects on the gonads were observed in rats and mice of both sexes after administration of phenol in the drinking water for 13 weeks (at approximately 819 and 380 mg/kg-bw per day for rats and mice, respectively; ATSDR, 1989) or for two years (at approximately 630 and 523 mg/kg-bw per day for rats and mice, respectively; ATSDR, 1989) (NCI, 1980). Stunted growth in young and a lack of reproduction were reported in rats administered very high concentrations of phenol (approximately 980 and 1680 mg/kg-bw per day; Health Canada, 1994) in drinking water in an early, inadequately documented multigeneration study (Heller and Pursell, 1938).

The developmental toxicity of phenol has been examined in several studies in rats and mice. In a well-conducted and well-documented study by Jones-Price *et al.* (1983a), a dose-related decrease in average fetal body weight per litter in CD rats exposed to 120 mg/kg-bw per day by gavage on days 6–15 of gestation was observed in the absence of maternal toxicity, based on clinical signs, maternal weight gain, maternal liver weight and gravid uterine weight. There were no effects at 60 mg/kg-bw per day, and there were no dose-related increases in mortality or structural malformations of the fetuses (offspring: NOAEL = 60 mg/kg-bw per day, LOAEL = 120 mg/kg-bw per day; mothers: NOEL = 120 mg/kg-bw per day). In contrast, there was no effect on numbers of implantation sites, resorptions, pre- and post-implantation loss, and numbers of live fetuses in Charles River rats at maternal doses of phenol of up to 180 mg/kg-bw per day on days 6–15 of gestation, although there were decreases in body weight gain in the mothers associated with decreases in food consumption (Procter and Gamble, 1993) (offspring: NOEL = 180 mg/kg-bw per day; mothers: LOEL = 120 mg/kg-bw per day). Narotsky and Kavlock (1995) reported a range of adverse perinatal effects in Fischer 344 rats exposed to 40 and 53 mg phenol/kg-bw per day on gestation days 6–19, including significantly reduced litter sizes and (at the higher dose) increased prenatal loss (offspring and mothers: LOAEL = 40 mg/kg-bw per day). Finally, hindlimb paralysis and/or short or kinky tails were observed in Sprague-Dawley rats exposed to 667 or 1000 mg/kg-bw on day 11 of gestation, but not at 333 mg/kg-bw (Kavlock, 1990) (offspring and mothers: NOEL = 333 mg/kg-bw per day,

LOAEL = 667 mg/kg-bw per day). In the last two studies, the developmental effects were accompanied by significant reductions in maternal body weight gain, although food consumption was not reported.

In the single well-conducted and well-documented study available in mice, a dose-related decrease in the average fetal body weight per litter was noted after the dams received up to 280 mg/kg-bw per day by gavage on days 6–15 of gestation (Jones-Price *et al.*, 1983b). This decrease was significant at the highest dose; however, this dose was clearly maternally toxic, based on observed mortality, clinical signs, reduced body weight and trend towards reduced liver weight (LOAEL = 280 mg/kg-bw per day; NOAEL = 140 mg/kg-bw per day). There was no convincing evidence of any teratogenic effects.

2.4.4 Effects in humans

2.4.4.1 Acute exposure and irritation

Phenol is acutely toxic to humans after both oral and dermal exposure, with children being more susceptible than adults (Deichmann, 1949; Evans, 1952; Lefaux, 1968; Griffiths, 1973; Lewin and Cleary, 1982; Allan, 1994). It is also severely irritating to the skin, eyes and mucous membranes (Abraham, 1972; Griffiths, 1973; Pardoe *et al.*, 1976; Schmidt and Maibach, 1981; Spiller *et al.*, 1993). Irritant dermatitis has resulted from application of topical medicines containing phenol (Fisher, 1980).

Phenol did not cause sensitization at a concentration of 2% in 25 volunteers after a series of five 48-hour exposures, each one preceded by a 24-hour pretreatment with sodium lauryl sulphate (Kligman, 1966).

2.4.4.2 Epidemiological studies

The potential carcinogenicity of phenol has been investigated in a small number of analytical epidemiological studies. Excesses of mortality from cancer of the lung were reported in a historical cohort study (Dosemeci *et al.*, 1991) and in a nested case–control study (Kauppinen *et al.*, 1993) at workplaces where phenol was used. There was also a non-significant trend between lung cancer mortality with increasing duration of exposure to phenol in a historical cohort study of workers in the wood industry with potential exposure to phenol (Blair *et al.*, 1990). However, the increase reported by Dosemeci *et al.* (1991) was mainly restricted to workers who were not exposed to phenol, while in the case–control study there was no clear trend in lung cancer mortality by duration of exposure (Kauppinen *et al.*, 1993). In addition, concentrations of phenol were not quantified in any of the studies, and in all instances there was potential exposure to a number of other substances (including formaldehyde, for which there is limited evidence for an association with lung cancer). With respect to other cancer sites, there was no consistent association between exposure to phenol and increased mortality from cancer in the available epidemiological studies (Wilcosky *et al.*, 1984; Dosemeci *et al.*, 1991; Siemiatycki, 1991; Pottern *et al.*, 1992; Kogevinas *et al.*, 1995).

Non-neoplastic effects have also been investigated in a few epidemiological studies. In a historical cohort and nested case–control study of rubber plant workers potentially exposed to phenol (exposure was defined simply as work in a process area when phenol was authorized for use), an increase in mortality from cardiovascular disease was observed, with some association with duration of exposure (Wilcosky and Tyroler, 1983). However, when workers exposed to other solvents were excluded from the analysis, numbers were too small to draw meaningful conclusions. In contrast, Dosemeci *et al.* (1991) observed declining trends in mortality from arteriosclerotic heart disease, as well as emphysema and cirrhosis of the liver, with increasing exposure to phenol (as cumulative exposure, duration of exposure or potential for dermal contact) in a historical cohort study of workers employed in five facilities that produced or used phenol or formaldehyde. In a small group of office workers exposed for six months to a mixture including phenol (approximately 1.3 mg/m^3 air), there were significant decreases

in lymphocyte subpopulations and suppression of proliferation of lymphocytes induced by phytohemagglutinin and alloantigens, compared with age- and sex-matched volunteers. Some of these effects were more pronounced, along with decreased erythrocyte counts and increased numbers of monocytes and eosinophils, in the subset of workers with the highest urinary phenol levels (Baj *et al.*, 1994). Hematological findings and increases in serum enzymes and electrolytes, of uncertain clinical significance, were reported by Shamy *et al.* (1994) in a small group of male workers exposed to 21.2 mg phenol/m^3.

2.5 Toxicokinetics/mechanism of action

Phenol is rapidly absorbed upon contact with skin and from the gastrointestinal tract and the lungs of both animals and humans. Once absorbed, it is rapidly distributed to the tissues and eliminated as metabolites, primarily in the urine (Capel *et al.*, 1972; Babich and Davis, 1981; Deichmann and Keplinger, 1981; Hiser *et al.*, 1994; Hughes and Hall, 1995). The metabolism of phenol occurs primarily by direct conjugation with glucuronic acid and sulphate in the intestine and liver, and to a lesser extent in other tissues. A small percentage of the absorbed dose of phenol is metabolized by cytochrome P450 enzymes to hydroquinone, which is then also subsequently conjugated with sulphate and glucuronic acid. The urinary metabolites of phenol that have been identified in mammals, including humans, are phenyl glucuronide, phenyl sulphate and the corresponding conjugates of hydroquinone — 4-hydroxyphenyl glucuronide and 4-hydroxyphenyl sulphate (Capel *et al.*, 1972; Hiser *et al.*, 1994; Hughes and Hall, 1995). Based on studies in rodents and non-human primates, sulphate conjugation predominates; however, at higher doses (above the range of 1–10 mg/kg-bw per day), there is saturation of sulphation, resulting in an increase in the proportion of urinary metabolites recovered as the glucuronide conjugate of hydroquinone (Mehta *et al.*, 1978; Weitering *et al.*, 1979; Koster *et al.*, 1981; Hiser *et al.*, 1994; Kenyon *et al.*, 1995). Available data also indicate that there are age-related differences in conjugation capacity, with that of immature rats being more limited, resulting in increased excretion of the glucuronide conjugate of hydroquinone in young animals (Heaton and Renwick, 1991).

The profile of toxicity of hydroquinone, the putatively toxic metabolite of phenol, is quite similar to that of the parent compound, with the former being more potent in the induction of similar effects. After long-term oral exposure in rats, renal toxicity is observed at doses of 25 mg hydroquinone/kg-bw per day and above. This compound is a more potent genotoxicant than phenol, being mutagenic in mammalian cells *in vitro* and clastogenic *in vivo* and inducing increases in primarily benign tumours in the kidney of male rats and liver of male and female mice. Developmental toxicity has been observed only at maternally toxic doses (IPCS, 1994b). Rats produce more hydroquinone than humans, eliminating 21% of ingested ^{14}C-phenol as conjugates of this metabolite, compared with 1% in humans (Capel *et al.*, 1972).

3.0 ASSESSMENT OF "TOXIC" UNDER CEPA

3.1 CEPA 11(a): Environment

The environmental risk assessment of a PSL substance is based on the procedures outlined in Environment Canada (1997a). Analysis of exposure pathways and subsequent identification of sensitive receptors are used to select environmental assessment endpoints (e.g., adverse reproductive effects on sensitive fish species in a community). For each endpoint, a conservative Estimated Exposure Value (EEV) is selected and an Estimated No-Effects Value (ENEV) is determined by dividing a Critical Toxicity Value (CTV) by an application factor. A conservative (or hyperconservative) quotient (EEV/ENEV) is calculated for each of the assessment endpoints in order to determine whether there is potential ecological risk in Canada. If these quotients are less than one, it can be concluded that the substance poses no significant risk to the environment, and the risk assessment is completed. If, however, the quotient is greater than one for a particular assessment endpoint, then the risk assessment for that endpoint proceeds to an analysis where more realistic assumptions are used and the probability and magnitude of effects are considered. This latter approach involves a more thorough consideration of sources of variability and uncertainty in the risk analysis.

3.1.1 Assessment endpoints

In Canada, most of the releases of phenol are to air and water. Since phenol is not persistent in the environment, environmental effects are most likely to occur near point sources. The major industry sectors identified are listed in Table 2. Releases from MWTPs were also included in this assessment.

When phenol is released to air, about half of it is found in air, while most of the remainder is found in soil. This is partially supported by air and soil measurements in Canada (Environment Canada, 1998a). Therefore, the assessment of phenol released to air focusses on terrestrial organisms exposed to air and soil near point sources.

When phenol is released to water, most of it is found in the water phase. Based on the sources and the physical and chemical properties of phenol, concentrations in sediment do not appear to be a concern. This is supported by many water and limited sediment measurements in Canada (Environment Canada, 1998a). Therefore, the assessment of phenol released to water focusses on organisms exposed in water near point sources.

3.1.1.1 Assessment endpoints for releases to water

Assessment endpoints include abundance and survival of fish, invertebrates, amphibians and algae. These organisms are an integral part of ecosystems, as each trophic level provides food for higher levels in the aquatic food chain. For example, algae are primary producers, forming the base of the food chain. Phytoplankton abundance and productivity are important in aquatic ecosystems because phytoplankton provide food for a variety of planktivorous organisms and thus control energy flow in a portion of the ecosystem. Cladocerans such as *Daphnia magna* consume bacteria and phytoplankton and are themselves consumed by many fish species. Various fish species feed on aquatic vegetation, phytoplankton, zooplankton, benthic invertebrates, benthic vertebrates, etc. Vertebrate omnivores provide food for vertebrate carnivores.

The most sensitive measurement endpoint identified for aquatic species was reproduction of rainbow trout measured as lethality of phenol to embryos and larvae.

3.1.1.2 Assessment endpoints for releases to air

The assessment endpoints chosen for terrestrial mammals were abundance and productivity of herbivores. These are important to preserve, since herbivores form important links in the food web as both predators and prey. Herbivores such as meadow voles (*Microtus pennsylvanicus*), which are small animals, are likely to have the highest exposure due to their rapid respiration rate and high metabolism. The most sensitive measurement endpoints identified for herbivores are inhalation and ingestion effects on rats.

Organisms such as plants and invertebrates near point sources are also likely to be exposed to phenol in air, rain and soil. Soil exposure can result from contact with phenol in soil particles, soil water and vapour. The assessment endpoints chosen for exposure to phenol in soil are the diversity, abundance and reproduction of terrestrial vegetation and invertebrates. Herbaceous vegetation is important because it provides food and/or habitat for terrestrial organisms as well as soil cover to reduce erosion and moisture loss. The most sensitive measurement endpoint identified for terrestrial plants was inhibition of seedling emergence of lettuce.

3.1.2 Environmental risk characterization

3.1.2.1 Aquatic organisms

Final effluent end-of-pipe concentrations were used as a measure of exposure of aquatic organisms because ambient water concentrations were not available near point sources. Recent concentrations reported for the pulp, paper and wood products, steel and metal products, and petroleum refining and products sectors, as well as for MWTPs, have been collected to reflect present exposures (Table 4).

TABLE 4 Hyperconservative assessment: environmental risk to rainbow trout

Sources[1]	Range of highest EEVs (mg/L)	CTV (mg/L)	Application factor	ENEV (mg/L)	Quotient <1	Quotient >1
Pulp, paper and wood products sector: **26 mills**	ND[2]–0.40 (phenol concentrations	0.01	10	0.001	11 mills	15 mills
Steel and metal products sector: **8 outfalls**[3]	0.006–0.34 (total phenolics concentrations)	0.01	10	0.001	0 outfall	8 outfalls
Petroleum refining and products sector: **16 refineries**	0.0004–2.03 (total phenolics concentrations)	0.01	10	0.001	1 refinery	15 refineries
MWTPs: **31 plants**	0.002–2.60 (total phenolics concentrations)	0.01	10	0.001	0 plants	31 plants

[1] EEVs for the 1) pulp, paper and wood products sector represent 1996 data, 2) steel and metal products sector represent 1995–1997 data, 3) petroleum refining and products sector represent 1993–1996 data, and 4) MWTPs represent 1985–1997 data. Data represent effluent concentrations.

[2] ND = not detected.

[3] Environmental assessment conducted on four mills that have a total of eight outfalls.

The CTV was selected from a data set composed of chronic studies conducted on several different types of organisms. The CTV is a chronic 27-day LC_{25} of 0.01 mg/L for the embryo-larval stage of rainbow trout (Birge *et al.*, 1979), the most sensitive aquatic species identified. For the hyperconservative assessment, the ENEV was derived by dividing the CTV of 0.01 mg/L by an application factor of 10. This factor accounts for the uncertainty surrounding the extrapolation of the LC_{25} to a long-term no-effects value, the extrapolation from laboratory to field conditions, and interspecies and intraspecies variations in sensitivity. The resulting ENEV is 0.001 mg/L.

In column 1 of Table 4, the sources and the number of sites per source (i.e., bolded text) considered in the hyperconservative assessment are presented. In column 2, the EEVs represent the highest monthly average concentration of phenol/total phenolics in final effluent for facilities in each source. For example, for the steel and metal products sector, eight EEVs are represented in the range of 0.006–0.34 mg/L. Monthly average concentrations were used because the ENEV was derived using a toxicity study conducted for a 27-day period. By doing so, exposure periods for the ENEV and EEVs are similar. For the MWTPs, monthly average concentrations were not available; EEVs represent the highest concentrations measured in 24-hour composites or grab samples. When phenol concentrations were not available, total phenolics concentrations were used as surrogates.

The results of the hyperconservative assessment are presented in Table 4 and demonstrate, for example, that the highest monthly average concentrations of total phenolics in 15 of the 16 petroleum refineries exceed the ENEV and produce a quotient greater than one. Because the analysis was hyperconservative, risks to aquatic biota populations are considered negligible for the refinery having a quotient below one. The hyperconservative assessment for refineries with a quotient equal to or greater than one proceeded to a conservative assessment. The same analysis was conducted for the other sources.

The toxicity data set on phenol is relatively large. The analysis of this data set showed that rainbow trout and leopard frog, in their early life stages, are at least two orders of magnitude more sensitive than other organisms to the effects of phenol. As a result, the application factor of 10 used in the hyperconservative assessment was reduced to 2 for the conservative assessment. The resulting conservative ENEV was 0.005 mg/L for lethality to the early life stages of rainbow trout.

The EEVs in the conservative assessment were derived by dividing the EEVs of the hyperconservative assessment by a generic and conservative dilution factor of 10 for all types of water bodies at all sites to estimate ambient concentrations near the outfalls (Appendix B of Environment Canada, 1998a). A further refinement of EEVs in the conservative assessment took into account the amount of phenol in total phenolics. When this amount was available for a particular source, EEVs were derived by dividing EEVs of the hyperconservative assessment by 10 (application of dilution factor) and multiplying by the estimated amount of phenol in total phenolics. The amount of phenol in total phenolics was estimated at 11% for the petroleum refining and products sector and 1% for the MWTPs (Environment Canada, 1998a).

Results of the conservative assessment are presented in Table 5 and demonstrate, for example, that the highest estimated ambient concentrations of phenol near the outfalls in 2 of the 15 petroleum refineries exceeded the ENEV and produced a quotient greater than one. Because the analysis was conservative, risks posed by phenol to aquatic biota populations are considered negligible for refineries having a quotient below one. The conservative assessment for the two refineries for which the quotient was equal to or greater than one proceeded to a probabilistic risk analysis. The same analysis was conducted for the other sources, with the result that the environmental assessment for four pulp, paper

TABLE 5 Conservative assessment: environmental risk to rainbow trout

Sources	Range of highest EEVs[1] (mg/L)	CTV (mg/L)	Application factor	ENEV (mg/L)	Quotient <1	Quotient >1
Pulp, paper and wood products sector: **15 mills**	0.00001–0.040 (phenol concentrations)	0.01	2	0.005	11 mills	4 mills
Steel and metal products sector: **8 outfalls**	0.0006–0.034 (total phenolics concentrations)	0.01	2	0.005	6 outfalls	2 outfalls
Petroleum refining and products sector: **15 refineries**	0.000 01–0.022[2] (phenol concentrations)	0.01	2	0.005	13 refineries	2 refineries
MWTPs: **31 plants**	0.000 002–0.003[3] (phenol concentrations)	0.01	2	0.005	31 plants	0 plants

[1] Estimated by assuming a dilution factor of 10.

[2] Also assumes 11% phenol in total phenolics.

[3] Also assumes 1% phenol in total phenolics.

and wood products mills and two steel and metal products outfalls proceeded to a probabilistic risk analysis.

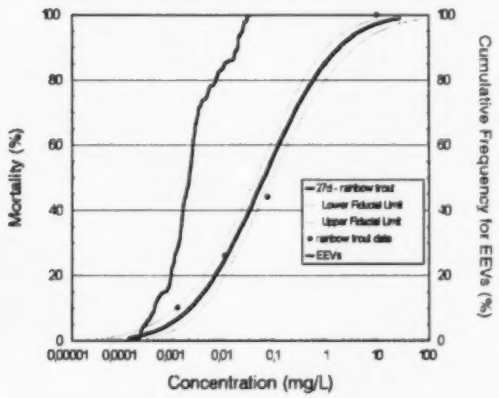
For the MWTPs, the environmental assessment ended at the conservative assessment. Aside from the data presented in Table 5, other exposure data for this source showed that in all cases phenol has a low probability of causing adverse effects on biota near outfalls of the MWTPs. The data used in this analysis included measured concentrations of phenol in final effluents for two MWTPs in Ontario in 1997 (phenol was not detected, detection limit 0.0017 mg/L) and concentrations of phenol in final effluents for seven MWTPs in Ontario in 1987 (maximum concentration 0.0173 mg/L) (Section 2.3.2.4).

A probabilistic risk analysis was conducted for the remaining sites of the three industry sectors. For details on each analysis and the raw data used, consult Environment Canada (1998a). The concentration–response curves for the embryo-larval stages of rainbow trout and leopard frog are very similar (Figure 2); thus, it can be expected that similar results for the probabilistic risk analysis would be obtained by using either curve. However, because the curve for rainbow trout indicates greater sensitivity at lower concentrations, it was used for this analysis.

Figure 3 shows the EEV distribution for the petroleum refining and products sector and the concentration–response curve for toxicity to the early life stages of rainbow trout. The EEVs were estimated by applying a dilution factor of 10 and a factor of 11% phenol in total phenolics to all of the 1995–96 monthly average final effluent concentrations (405 values) for the two refineries whose assessment proceeded to the probabilistic risk analysis. Because of the availability of the data, the 405 values (range 0.0001–0.0312 mg/L) were calculated as a monthly running average (e.g., January 1 to January 31, January 2 to February 1, etc.). The exposure and effect distributions were combined to estimate the probabilities of effects of differing magnitude. This was done by calculating effects doses from the function shown in Figure 3 from responses ranging from 1 to 99% and by comparing these effects doses with the cumulative distribution function for concentration (Figure 3). Combining these two distributions generated the risk curve

for chronic toxicity to rainbow trout. Figure 4 shows, for example, that there is a 41% probability of at least 15% mortality to the early life stages of rainbow trout exposed to phenol from releases of effluents from the two worst-case refineries in Canada. The results indicate higher probabilities (84%) of minor effects (>3% mortality) and low probabilities (2%) of more serious effects (>35% mortality). The results of the probabilistic risk analysis for the pulp, paper and wood products sector and the steel and metal products sector are very similar and are presented in Figures 5 and 6, respectively.

FIGURE 3 Probabilistic risk analysis: exposure distribution for two refineries using 1995–96 monthly average concentrations of phenol (EEVs) and effect distribution for 27-day lethality to embryo-larval stage of rainbow trout



As part of a weight-of-evidence for the probabilistic risk analysis, chronic effects of phenol on aquatic communities were also estimated using the Water Environment Research Foundation approach (Parkhurst *et al.*, 1996). In this approach, a logistic regression model was fit to the cumulative frequency of species acute endpoints (e.g., LC_{50}) and phenol concentration. Because there were insufficient empirical chronic toxicity data to estimate effects on aquatic communities chronically exposed to phenol, the acute model estimates were divided by an acute to chronic ratio equal to 6.67 (Hodson *et al.*, 1991).

FIGURE 4 Probabilistic risk analysis: risk distribution for chronic toxicity to the embryo-larval stage of rainbow trout for two refineries in 1995–96

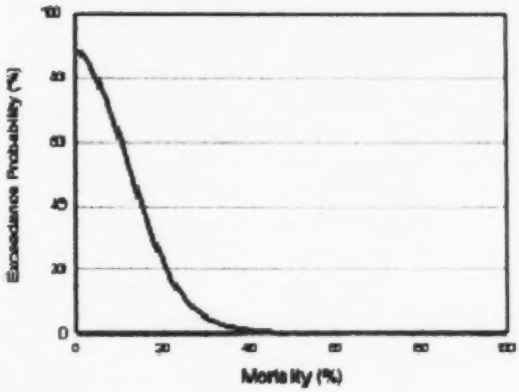


FIGURE 5 Probabilistic risk analysis: risk distribution for chronic toxicity to the embryo-larval stage of rainbow trout for four pulp, paper and wood products mills in 1996

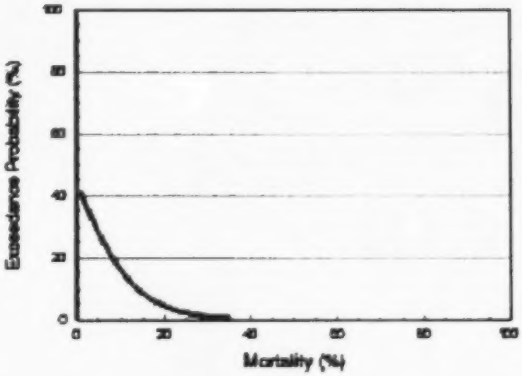


Figure 7 shows the acute and chronic distributions for toxicity endpoints of various freshwater and marine species and the distribution of EEVs (1995–96) for the two refineries. Combining the distributions of chronic effects and EEVs generated the risk curve in Figure 8, which shows a very low probability of minor effects at the community level of organization. The results of this approach for the pulp, paper and wood products sector and the steel and metal products

FIGURE 6 Probabilistic risk analysis: risk distribution for chronic toxicity to the embryo-larval stage of rainbow trout for two steel and metal products mills in 1995–1997

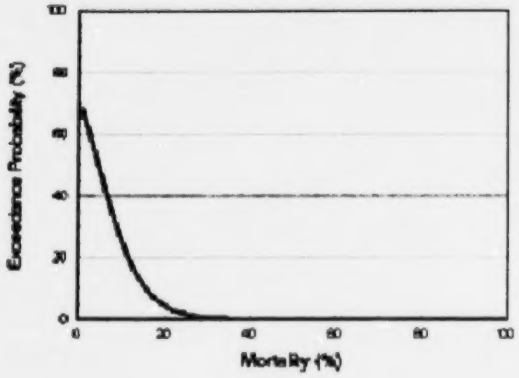
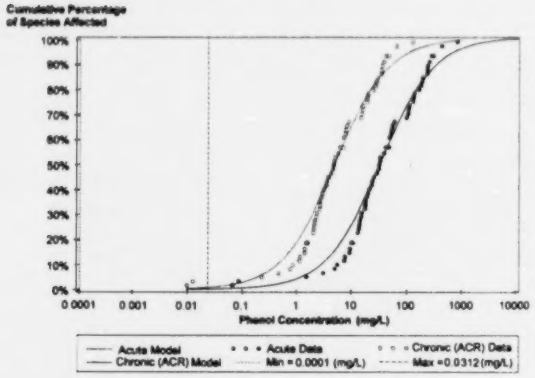


FIGURE 7 Probabilistic risk analysis: aquatic community risk model results for acute and chronic toxicity compared with the range of EEVs for two petroleum refineries in 1995–96

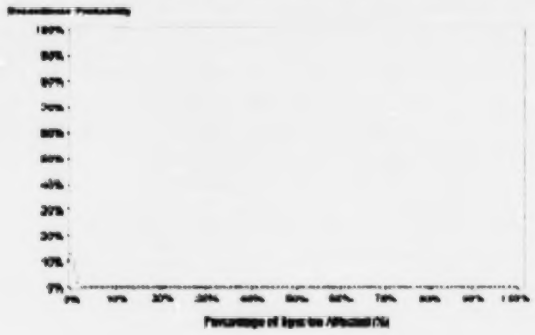


sector are very similar and are presented in Environment Canada (1998a).

3.1.2.1.1 Discussion

In this section, other lines of evidence with regard to the industrial outfalls that could pose a risk to aquatic biota are discussed.

FIGURE 8 Probabilistic risk analysis: risk distribution for chronic toxicity at the community level of organization for two petroleum refineries in 1995–96



Exposure

Several lines of evidence suggest that exposure was likely overestimated in some or all of the risk scenarios considered in Section 3.1.2.1. For example, it was assumed for each of the eight steel mill outfalls that the concentration of phenol was equal to the concentration of total phenolics. This is a very conservative assumption, given that the limited available data for other sources to the aquatic environment indicate that the percentage of phenol in total phenolics is generally low (average 11% for effluents from petroleum refining and products plants, 1% for effluents from MWTPs). No information was available, however, to estimate the proportion of phenol in total phenolics in effluents from steel mills.

Examination of the raw data for monthly effluent concentrations for each of the industrial sectors considered indicated that phenol levels have been declining in recent years (Environment Canada, 1998a). Therefore, current and future levels are likely to be lower than the levels used to estimate risks in the hyperconservative assessment through the probabilistic risk analysis. Further, the effluent monitoring data indicated that the distributions of phenol levels are almost always right skewed (i.e., few high values and the majority of low values). Therefore, exceedances

of effect levels for sensitive biota were often due to short-term excursions well above typical concentrations in the effluent. If such exceedances were small, as the data indicate, then it is likely that many systems could recover from the chemical stress within a matter of weeks (Stephan *et al.*, 1985).

Effects

The risk analysis clearly indicates that only the most sensitive life stages of a very few sensitive aquatic biota (specifically rainbow trout and leopard frog; Figure 2) could be adversely affected by phenol at the levels found near several outfalls. The available effects data indicate that effects on early life stages of rainbow trout and leopard frogs occur at levels approximately two orders of magnitude below levels affecting most other aquatic biota (Figure 7).

Given that the concern is solely for early life stages of sensitive aquatic species, it is likely that elevated concentrations of phenol are only a concern during the period between egg laying and early growth of fish larvae. For rainbow trout in most regions of Canada, spawning takes place from mid-April to late June, with eggs hatching 4–7 weeks later (Scott and Crossman, 1973). Fry usually commence feeding 15 days after hatching and emerge from the nests from mid-June to mid-August. For leopard frog, spawning takes place from April to May, with eggs hatching within 2 weeks (Froom, 1982). Examination of the estimated ambient concentrations near the outfalls (EEVs for the probabilistic risk analysis) indicates that only 11% of these concentrations exceeded the CTV (27-day LC_{25} of 0.01 mg/L) during this time frame. The exceedance value of 11% was derived using data for a few years only and may therefore not reflect exposures over a longer period of time.

High mortality during early life stages is typical for most aquatic species, including salmonids and amphibians (Power and Power, 1994, 1995). Given this high natural mortality, the question is: What does a modest increase in mortality due to exposure to phenol mean for a population as a whole? To answer this question would require population-level modelling and field studies, which are beyond the scope of this assessment. However, several studies have shown that a modest increase in mortality of early life stages may be mitigated by a number of compensatory phenomena. For example, increased mortality from exposure to a contaminant may be offset by increased survival and fecundity of remaining individuals due to reduced density dependence (Ferson *et al.*, 1996). Ferson *et al.* (1996) showed that brook trout (*Salvelinus fontinalis*) do not experience population declines when under moderate stress (e.g., 20% decline in fecundity), but populations crash after some breaking point (e.g., 75% decline in fecundity). Given that probabilities of effects greater than 35% (mortality) on early life stages of rainbow trout and leopard frogs are low (<10%), it seems unlikely that populations near industrial outfalls emitting phenol will experience severe declines.

3.1.2.1.2 Summary

In summary, the hyperconservative assessment through the probabilistic risk analysis indicated that:

- Risks to early life stages of sensitive aquatic biota are negligible for 22/26 pulp and paper mills in Ontario, 6/8 steel mill outfalls in Canada, 14/16 petroleum refining and product plants in Canada, and all 31 MWTPs in Canada for which data were available.
- Of the remaining plants, the probability of effects greater than 35% for early life stages of the most sensitive species exposed to phenol near outfalls was low (<5%).
- The probability of phenol levels near outfalls causing effects on greater than 5% of aquatic communities was negligible for all pulp and paper mills, steel mills, petroleum refining and product plants and MWTPs considered in this assessment.

Therefore, the results of the hyperconservative assessment through the probabilistic risk analysis

and other lines of evidence discussed above indicate that phenol from industrial and municipal point sources poses little risk to aquatic organisms in Canada.

3.1.2.2 Terrestrial organisms

The worst-case concentration in air nearest to Company 1 was predicted by the SCREEN3 model, to be four orders of magnitude greater than those of Companies 2 and 3. As a result, the focus of this assessment was to investigate the potential for effects resulting from releases from Company 1.

3.1.2.2.1 Mammals

Effects on herbivores such as meadow vole were estimated using two exposure scenarios: 1) air only, and 2) air and food. No effects data have been identified for herbivores. As a result, effects of phenol on rats were used to establish the CTV.

For scenario 1, the CTV is 98 mg/m^3, based on a 14-day NOAEL (highest concentration tested; "unbounded") for inhalation by rats (CMA, 1998). For the hyperconservative assessment, the ENEV was derived by dividing the CTV by an application factor of 100. This factor accounts for the limited data set on inhalation effects of phenol on mammals, the uncertainty surrounding the extrapolation from laboratory to field conditions, interspecies and intraspecies variations in sensitivity and extrapolation from a subchronic to a long-term no-effects value. As a result, the ENEV was 0.98 mg/m^3.

The EEV used in the hyperconservative assessment was the highest predicted 24-hour average air concentration of 0.022 mg/m^3 for the nearest field outside the boundaries to the highest emitter of phenol to air in Canada, Company 1. This concentration was predicted using the air dispersion model ISCST3, which was applied to simulate a 5.5-year period using meteorological data. A 24-hour concentration was selected because it provides a more reasonable conservative exposure concentration than values predicted for shorter time periods (e.g., 1 hour).

The hyperconservative quotient was calculated as follows.

$$\text{Quotient} = \frac{\text{EEV}}{\text{ENEV}}$$

$$= \frac{0.022 \text{ mg/m}^3}{0.98 \text{ mg/m}^3}$$

$$= 0.02$$

The hyperconservative quotient is less than one. As a result, it is unlikely that phenol will cause chronic adverse effects via airborne exposure on herbivores living in the field closest to the highest emitter of phenol in Canada. Since the worst-case ground-level air concentration predicted using the SCREEN3 model was four orders of magnitude higher near Company 1 than near the second and third highest emitters, it is unlikely that phenol will cause chronic adverse effects via airborne exposure on herbivores living near any other point source in Canada.

As part of a weight-of-evidence, a hyperconservative quotient was also calculated using the highest measured concentration of phenol in air in Canada of 0.476 mg/m^3 detected in the vicinity of a train carrying treated wood (Strosher, 1982). Although the focus of this assessment was not to include the pesticidal use of phenol, a quotient of 0.5 was obtained when this concentration was used as an EEV. This result also suggested that phenol is not likely to cause adverse effects via this exposure route.

The use of an "unbounded" NOAEL, the use of a NOAEL instead of a LOAEL and the estimation of a 24-hour air concentration instead of a 14-day concentration are factors that contribute to the conservatism of this hyperconservative assessment, indicating that phenol is even less likely to cause adverse effects via this exposure route. There was no need to proceed to a conservative assessment for this scenario.

For scenario 2, the Wildlife Contaminant Exposure Model (WCEM), developed by the Canadian Wildlife Service of Environment Canada, was used to estimate a total daily phenol intake (i.e., EEV) for meadow vole in the field nearest to Company 1 through uptake from air and food (e.g., shoots). Using WCEM, a daily inhalation rate equal to 0.11 m^3/day was calculated using the allometric equation (1) derived by Stahl (1967) (using a correction factor of 3 as recommended by U.S. EPA, 1993, to convert from standard to field metabolic rates) and a mean body weight of 33.4 g for a female meadow vole (Brochu *et al.*, 1988):

$$\text{daily inhalation rate } (m^3/\text{day}) = 0.002\,171 \times \text{weight } (g)^{0.8} \quad (1)$$

A summer weight was chosen to reflect spring/summer conditions, as these are the seasons when exposure via a plant diet is most likely to occur. The daily inhalation rate (0.11 m^3/day) was then multiplied by the highest predicted 24-hour average concentration of 0.022 mg/m^3 for the field nearest to the highest emitter of phenol to air in Canada, which resulted in an intake via the inhalation route, as calculated by the WCEM, of 0.071 mg/kg-bw per day.

The WCEM was also used to calculate the daily intake via ingestion of food, first by calculating a free-living metabolic rate using the allometric equation (2) derived by Nagy (1987):

$$\text{free-living metabolic rate (kcal/day)} = 2.514 \times \text{weight } (g)^{0.507} \quad (2)$$

where the free-living metabolic rate includes energy expenditures such as basal metabolic rate, thermoregulation, locomotion, feeding, predator avoidance, alertness, posture, etc. The free-living metabolic rate was calculated by the WCEM to be 14.8 kcal/day.

To estimate a daily intake rate via the ingestion route, a diet intake rate and the concentration of phenol in the diet must be determined. Summer food habits of the meadow vole in a grassy habitat have been analysed by Lindroth and Batzli (1984). In this study, a typical diet of a meadow vole, based on stomach contents, was determined to be 65% dicot shoots (leafy plants), 29% monocot shoots (grasses or grains), 4% insects, 1% seed and 1% fungi. Using the proportions of this diet, free-living metabolic rate, gross energies and assimilation efficiencies, and applying the methodology described in U.S. EPA (1993), a diet intake rate of 22.4 g/day was calculated using the WCEM (Appendix C of Environment Canada, 1998a).

The maximum daily total deposition of phenol in the field nearest (750 m from the stack) to the highest emitter of phenol was predicted using the air dispersion model ISCST3 over a 5.5-year period (Davis, 1997). The maximum daily total deposition was estimated to be 44.5 mg phenol/m^2. Whittaker (1975) estimated the dry weight biomass per unit area for temperate grassland to range from 0.2 to 5 kg plant/m^2, with a mean value of 1.6 kg plant/m^2 (dry weight). Assuming that grass and leafy vegetation are approximately 80% water, a wet weight biomass per unit area ranging from 1 to 25 kg plant/m^2 and a mean value of 8 kg plant/m^2 were calculated. The most conservative estimate of concentration of phenol in vegetation was calculated by dividing the total daily deposition of phenol, which is assumed to fall on and be assimilated by vegetation in the field, by the minimum wet weight biomass of the range presented above. The resulting concentration of phenol in vegetation was:

$$\frac{\text{total daily deposition}}{\text{minimum wet weight biomass}} = \frac{44.5 \text{ mg phenol/m}^2}{1 \text{ kg plant/m}^2}$$

$$= 44.5 \text{ mg phenol/kg plant}$$

The daily concentration of 44.5 mg phenol/kg plant was used to estimate the steady-state concentration in vegetation using the following equation (Mackay, 1991):

$$\frac{\text{daily concentration in vegetation}}{(0.693/\text{half-life of phenol in vegetation})}$$

$$= \frac{44.5 \text{ mg/kg plant per day}}{(0.693/2.3 \text{ days})}$$

$$= 148 \text{ mg/kg plant}$$

where "0.693/half-life of phenol in vegetation" is reciprocal days and the "half-life of phenol in vegetation" is assumed to be equal to the half-life of phenol in water (55 hours). The concentration of phenol in vegetation at equilibrium via air–plant exchange was also estimated using partition coefficients and assuming 2% lipid in vegetation. This method resulted in concentrations of phenol a few orders of magnitude lower than that calculated by the deposition method presented above. As a result, the deposition method was preferred, because it was more conservative.

The concentration of 148 mg phenol/kg plant and 94% of the diet intake rate of 22.4 g/day (94% equals diet of shoots only; remainder of diet is assumed to have negligible amounts of phenol) were used as input in the WCEM, which estimated a daily intake via the ingestion route of 93 mg phenol/kg-bw per day (Appendix D of Environment Canada, 1998a).

The total daily intake of phenol (EEV) is obtained by adding the intake rates via the inhalation and ingestion routes, which gives an EEV of 93 mg phenol/kg-bw per day. This EEV assumes that all phenol deposited on vegetation was available for uptake by meadow vole and was not excreted or metabolized before any effects occurred during that one-day exposure.

The CTV is 40 mg/kg-bw per day, based on a LOAEL (lowest concentration tested; "unbounded") for the significant reduction of litter sizes for rats (Narotsky and Kavlock, 1995). (Although the maternal and fetal toxicities observed at this dose were not confirmed in other studies of developmental effects in rats, this study was selected because it yields the most conservative assessment for this hyperconservative scenario, and the reductions in litter size are clearly relevant to population-level effects.) For the hyperconservative assessment, the ENEV was derived by dividing the CTV by an application factor of 20. This factor accounts for the uncertainty surrounding the extrapolation from laboratory to field conditions, the extrapolation from a LOAEL to a long-term no-effects value, and interspecies and intraspecies variations in sensitivity. As a result, the ENEV was 2 mg phenol/kg-bw per day.

The hyperconservative quotient was calculated as follows:

$$\text{Quotient} = \frac{\text{EEV}}{\text{ENEV}}$$

$$= \frac{93 \text{ mg/kg-bw per day}}{2 \text{ mg/kg-bw per day}}$$

$$= 46.5$$

Because the hyperconservative quotient exceeds one, the environmental assessment for this scenario proceeded to a conservative assessment.

As noted in Sections 2.3.2.1 and 2.3.2.7, the distributions of predicted air concentrations and soil depositions are right skewed (i.e., few occurrences with high concentrations). The maximum daily total deposition of 44.5 mg phenol/m^2 and the maximum 24-hour concentration of phenol in air of 0.022 mg phenol/m^3 were used in the hyperconservative assessment to estimate exposure in the field nearest to Company 1. These values, predicted by the ISCST3 model, occur for only one day during a 5.5-year period. These values were purposely used in the hyperconservative assessment to overestimate exposure and generate a very conservative scenario. In the conservative assessment, these values were revised to provide a more realistic exposure over that time period. The values used in the conservative assessment were the medians of the distributions, equal to 7×10^{-6} mg/m^3 for the concentration of phenol in air and 3.05 mg/m^2 for the deposition to soil.

Medians were used because they represent the most typical values to occur at any given time and are therefore most appropriate for deriving steady-state concentrations, which assumes constant deposition over a several-week period. Also, in the conservative assessment, the mean wet weight biomass of 8 kg/m² was used to provide a more realistic estimate of the concentration of phenol in vegetation. Based on these revised values, the concentration of phenol in vegetation is estimated to be:

$$\frac{\text{total daily deposition}}{\text{mean wet weight biomass}}$$

$$= \frac{3.05 \text{ mg phenol/m}^2}{8 \text{ kg plant/m}^2}$$

$$= 0.38 \text{ mg phenol/kg plant}$$

The daily concentration of 0.38 mg phenol/kg plant was used to estimate the steady-state concentration in vegetation using the following equation (Mackay, 1991):

$$\frac{\text{daily concentration in vegetation}}{(0.693/\text{half-life of phenol in vegetation})}$$

$$= \frac{0.38 \text{ mg/kg plant per day}}{(0.693/2.3 \text{ days})}$$

$$= 1.3 \text{ mg/kg plant}$$

where "0.693/half-life of phenol in vegetation" is reciprocal days and the "half-life of phenol in vegetation" is assumed to be equal to the half-life of phenol in water (55 hours).

The concentration of 1.3 mg phenol/kg plant, the median 24-hour concentration in the field of 7×10^{-6} mg/m³, the daily inhalation rate of 0.11 m³/day and 94% of the daily diet intake of 22.4 g/day were used as input in the WCEM, which gave a multimedia EEV of 0.82 mg phenol/kg-bw per day (Appendix E of Environment Canada, 1998a).

The ENEV used in the hyperconservative assessment was also used in the conservative assessment. The conservative quotient was calculated as follows:

$$\text{Quotient} = \frac{\text{EEV}}{\text{ENEV}}$$

$$= \frac{0.82 \text{ mg/kg-bw per day}}{2 \text{ mg/kg-bw per day}}$$

$$= 0.4$$

The conservative quotient is less than one. As a result, there is a low probability of effects on herbivores exposed to phenol via air and ingestion in the nearest field of the worst-case emitter of phenol to air in Canada. Since the worst-case ground-level air concentration predicted using SCREEN3 is four orders of magnitude higher near Company 1 than near the second and third highest emitters, it is unlikely that phenol will cause chronic adverse effects via airborne exposure on herbivores living near any point source in Canada.

The approach used to estimate the maximum concentration of phenol in vegetation and meadow vole assumes that 100% of the phenol was deposited on vegetation and entered the plants. This assumption is conservative. Therefore, it is likely that the concentration of phenol consumed by meadow vole through vegetation is overestimated. In addition, the ENEV also contributes to the conservatism of this assessment, since in other studies in rats, maternal and/or fetal toxicity were observed only at doses higher than the LOAEL in the study on which the CTV is based. As a result, it is unlikely that phenol causes adverse effects on herbivores near point sources in Canada. There was no need to proceed to a probabilistic risk analysis.

3.1.2.2.2 Terrestrial vegetation

For exposure of terrestrial vegetation through soil, the CTV was the chronic five-day EC_{25} of 79 mg/kg dry weight for inhibition of seedling emergence for lettuce (Environment Canada, 1995). For the hyperconservative assessment, the ENEV was derived by dividing the CTV by an application factor of 10. This factor accounts for the uncertainty surrounding the extrapolation from an EC_{25} to a long-term no-effects value, the extrapolation from laboratory to field conditions, and interspecies and intraspecies variations in sensitivity. The resulting ENEV was 7.9 mg/kg dry weight.

In 1995, the highest concentration of phenol in surface soil in Canada was measured on site at the Company 1 plant. The concentration was 1.7 mg/kg dry weight (Géologos Inc., 1997). This value was therefore used as the EEV in the hyperconservative assessment.

The hyperconservative quotient was calculated as follows:

$$\text{Quotient} = \frac{\text{EEV}}{\text{ENEV}}$$

$$= \frac{1.7 \text{ mg/kg dry weight}}{7.9 \text{ mg/kg dry weight}}$$

$$= 0.2$$

The hyperconservative quotient for toxicity to terrestrial vegetation is lower than one. Based on this result, it is unlikely that phenol will cause chronic adverse effects via soil exposure on terrestrial vegetation near point sources in Canada. There was no need to proceed to a conservative assessment.

3.2 CEPA 11(b): Environment upon which human life depends

Once released in the atmosphere, phenol is likely to be removed via photooxidation by reaction with hydroxyl and nitrate radicals, photolysis and wet precipitation (Atkinson *et al.*, 1987, 1992; Bunce, 1996). Because of its reactivity in the atmosphere, the Photochemical Ozone Creation Potential is substantial; however, quantities available for reaction make the contribution insignificant relative to those of other smog-forming substances. Reaction with ozone is negligible, and the absence of chlorine and bromine atoms in the molecule and the overall short half-life of phenol mean that its potential contributions to stratospheric ozone depletion and climate change are both negligible (Bunce, 1996).

3.3 CEPA 11(c): Human health

3.3.1 Estimated population exposure

Data on levels of phenol in environmental media in Canada to serve as the basis for development of estimates of population exposure are limited to very few samples of ambient air removed from point sources in early studies in other countries and to limited Canadian surveys in drinking water, soil and food in which phenol was seldom detected. Meaningful probabilistic exposure assessment is precluded, therefore, and in this section, mean deterministic estimates of environmental exposure (from air, water, soil and food) by members of the general population of Canada have been derived. Worst-case intakes have also been estimated for populations near industrial point sources of phenol, although, given the limitations of available data, these are upper-bound estimates that, in all likelihood, overestimate exposure. Finally, intakes have been estimated for persons using consumer products that can result in exposure to phenol (cigarettes, certain non-prescription drugs).

Point estimates of average daily intake (on a body weight basis), based on the data for ambient air, drinking water, soil and food summarized in Section 2.3.2 and reference values for body weight, inhalation volume and amounts of drinking water, soil and food ingested daily, are presented for six discrete age groups of the Canadian general population in Table 6. Based on these estimates, the total daily estimated intake of phenol for the general non-smoking population ranges from approximately 0.06 to 0.71 µg/kg-bw per day. (The range of estimates primarily reflects the bounding estimates from these surveys, calculated by assuming, in separate calculations, either zero or the detection limit for those samples in which phenol was not detected.)

Although somewhat uncertain, based on these estimates, ingestion of food is likely the principal route of exposure to phenol for non-smoking members of all age groups in populations removed from point sources. This is the case even when concentrations of phenol are assumed to be zero in foodstuffs in which it was not detected and is consistent with the physical/chemical properties of phenol, which indicate that it is not volatile, and the results of fugacity modelling (Section 2.3.1.6). In addition, phenol is expected to be present in foods of animal origin as a result of endogenous production by microflora in the gut. Exposure from ingestion of drinking water and soil appears to be negligible compared with that from food.

Exposures from ambient air may be substantially higher for populations located in the vicinity of some point sources. Based on the upper end of the range of concentrations reported in the vicinity of industrial plants in Alberta and Ontario (Section 2.3.2.1), and assuming the same reference values for inhalation volume and body weight as for the general population, the maximum intake of phenol from ambient air in the vicinity of point sources is estimated to range from 12 to 34 µg/kg-bw per day for the various age classes (Table 7). A "worst-case" or upper-bound estimate for exposure from ambient air near a point source was also calculated, using the maximum 24-hour concentration of phenol (145 µg/m^3) predicted by air dispersion modelling of the facility that is the largest Canadian source of phenol releases to air (Davis, 1997). Based on this value, the estimated intake of phenol in ambient air near this point source ranges from 29 to 86 µg/kg-bw per day for the various age classes (Table 7).

The general population is also exposed periodically to phenol through the use of several consumer products, including, for example, mouthwashes, throat lozenges and sprays, and antiseptic lotions; complete quantitative estimation of exposure from such products is not possible due to the lack of adequate data on per capita use of such products. However, based on data available, the contribution to total phenol exposure from use of consumer medicinal products may be greater than that from environmental exposures. By way of example, estimates of potential intakes of phenol from the use of several consumer products were derived (Table 8). The estimates were generally based on the assumption that the product is used at the maximum dose recommended on the packaging. The estimated intakes are 4 µg/kg-bw per day for a medicated lip ointment, 29 and 81 µg/kg-bw per day for two brands of ointment, 3400 µg/kg-bw per day for a brand of sore throat lozenges, and 3300 µg/kg-bw per day for a sore throat spray. It is noted that those products for which the estimated intakes are highest (sore throat lozenges and sprays) are intended for occasional short-term use only. Mean intakes of phenol from cigarette smoking are also estimated to be elevated, ranging from 2.5 to 46 µg/kg-bw per day, depending on the content of phenol in mainstream smoke (Table 6). These estimates have been made for only a subset of the products through which the public may be exposed to phenol, but they serve to confirm that intakes from some products can be considerably higher than intakes from environmental exposures.

In addition to exposure of humans to exogenous phenol in environmental media, phenol is produced in the metabolism of tyrosine by

TABLE 6 Estimated daily intake of phenol via environmental media for the general population in Canada

Media	Estimated daily phenol exposure (µg/kg-bw per day)					
	0–0.5 years[1]	0.5–4 years[2]	5–11 years[3]	12–19 years[4]	20–59 years[5]	60+ years[6]
Ambient air[7]	0.033	0.072	0.056	0.032	0.028	0.024
Drinking water[8]	0–0.000 11	0–0.000 05	0–0.000 05	0–0.000 03	0–0.000 02	0–0.000 02
Soil[9]	0–0.000 39	0–0.000 64	0–0.000 21	0–0.000 05	0–0.000 04	0–0.000 04
Food[10]	0.028–0.68	0.20–0.59	0.20–0.45	0.19–0.32	0.49–0.58	0.32–0.40
Total	0.06–0.71	0.27–0.66	0.26–0.51	0.22–0.35	0.52–0.61	0.34–0.42
Cigarettes smokers[11]	–	–	–	–	2.5–46	2.5–46

[1] Assumed to weigh 7.6 kg, breathe 2.1 m³ of air per day, drink 0.2 L of tap water (excluding tap water-based beverages) per day and ingest 30 mg of soil per day (EHD, 1997).

[2] Assumed to weigh 15.6 kg, breathe 9.3 m³ of air per day, drink 0.2 L of tap water (excluding tap water-based beverages) per day and ingest 100 mg of soil per day (EHD, 1997).

[3] Assumed to weigh 31.2 kg, breathe 14.5 m³ of air per day, drink 0.4 L of tap water (excluding tap water-based beverages) per day and ingest 65 mg of soil per day (EHD, 1997).

[4] Assumed to weigh 59.7 kg, breathe 15.8 m³ of air per day, drink 0.4 L of tap water (excluding tap water-based beverages) per day and ingest 30 mg of soil per day (EHD, 1997).

[5] Assumed to weigh 70.7 kg, breathe 16.2 m³ of air per day, drink 0.4 L of tap water (excluding tap water-based beverages) per day and ingest 30 mg of soil per day (EHD, 1997).

[6] Assumed to weigh 70.6 kg, breathe 14.3 m³ of air per day, drink 0.4 L of tap water (excluding tap water-based beverages) per day and ingest 30 mg of soil per day (EHD, 1997).

[7] Since no adequate data concerning background concentrations of phenol in ambient air in Canada were located, these values are based on the mean concentration of phenol in ambient air of 0.12 µg/m³ reported in Jones (1976) for an urban/suburban site in the United States. Since no adequate data on levels in indoor air in North America were identified, it was assumed that indoor air levels were the same as ambient (outdoor) levels.

[8] Based on a survey by Sithole and Williams (1986) in which phenol was not detected at a quantitation limit of 0.004 µg/L in 120 samples of treated water from facilities across Canada (the range of estimated intakes was calculated assuming concentrations of phenol from 0 to 0.004 µg/L [the quantitation limit]).

[9] Based on the detection limit of 0.1 mg/kg in a survey by Golder Associates (1987), in which phenol was not detected in a limited survey of soils in residential and parkland areas in Ontario removed from point sources (the range of estimated intakes was calculated assuming concentrations of phenol from 0 to 0.1 mg/kg [the detection limit]).

[10] Based on the levels of phenol reported in 33 food composites collected in a market basket survey in Windsor, Ontario, in January 1992 (ETL, 1992) and the average daily consumption of the food group by each age group (NHW, 1977). In the market basket survey, phenol was present in cured pork (0.13 µg/g), organ meats (0.81 µg/g), cold cuts (0.08 µg/g), canned weiners (0.32 µg/g), cooking fats/salad oils/margarine (0.073 µg/g), tea/coffee (0.014 µg/g) and alcoholic drinks (0.13 µg/g). Phenol was not detected (detection limit 0.005 µg/g in solids, 0.0005 µg/g in liquids) in milk/cream/yogurt, cheese/butter, beef products, fresh pork, lamb products, chicken/turkey, eggs, fresh/frozen marine fish, fresh/frozen freshwater fish, canned fish, shellfish, canned meat soup, canned vegetable soup, dehydrated soup, bread/rolls/biscuits, flour/cake/cookies/danish + donuts/muffins/pancakes/crackers/pizza, breakfast cereals, pies, pasta, root vegetables, non-root vegetables, fresh/frozen/dried fruit, canned/bottled/juice fruit, peanuts/peanut butter, sugar/jam products and soft drinks. The range of estimated intakes was calculated assuming all non-detectable concentrations equal to zero (lower end of range) and equal to the detection limit (upper end of range). Since no data were identified on levels of phenol in breast milk, it was assumed that infants consumed table foods.

[11] Based on the range of values for the content of phenol in mainstream smoke (9–161 µg per cigarette) reported in ATSDR (1989) and consumption of 20 cigarettes per day, the approximate mean number smoked by regular Canadian smokers aged 15 years or older as of 1995 (Kaiserman, 1997).

TABLE 7 Worst-case estimates of daily intake of phenol from inhalation of air by populations in the vicinity of point sources in Canada

Data used	Estimated daily phenol exposure (µg/kg-bw per day)					
	0–0.5 years[1]	0.5–4 years[2]	5–11 years[3]	12–19 years[4]	20–59 years[5]	60+ years[6]
Measured level[7]	16	34	26	15	13	12
Dispersion modelling[8]	40	86	67	38	33	29

[1] Assumed to weigh 7.6 kg and breathe 2.1 m³ of air per day (EHD, 1997).

[2] Assumed to weigh 15.6 kg and breathe 9.3 m³ of air per day (EHD, 1997).

[3] Assumed to weigh 31.2 kg and breathe 14.5 m³ of air per day (EHD, 1997).

[4] Assumed to weigh 59.7 kg and breathe 15.8 m³ of air per day (EHD, 1997).

[5] Assumed to weigh 70.7 kg and breathe 16.2 m³ of air per day (EHD, 1997).

[6] Assumed to weigh 70.6 kg and breathe 14.3 m³ of air per day (EHD, 1997).

[7] Based on the upper end of the range of measured concentrations of phenol in very limited short-term surveys of ambient air from the vicinity of industrial plants (wood treatment plants, phenol-formaldehyde resin plants and a rape seed plant) in Alberta and Ontario, which ranged from not detected to 57 µg/m³ (Strosher, 1982; De Brou and Bell, 1987; De Brou and Ng, 1989; De Brou, 1990). In the absence of data on levels of phenol in indoor air in these populations, indoor air levels were assumed to be the same as ambient (outdoor) levels.

[8] Based on the maximum 24-hour concentration of phenol (145 µg/m³) predicted by air dispersion modelling at the facility that is the largest Canadian source of phenol releases to air (Davis, 1997). In the absence of data on levels of phenol in indoor air near this location, indoor air levels were assumed to be the same as ambient (outdoor) levels. There are also limited data on concentrations of phenol in soil from this location, but estimated exposure from this medium is negligible compared with that from air and is not included in the above estimates.

bacteria in the gut. This production is proportional to the amount of protein consumed and is in the range of 1–10 mg/day.

3.3.2 Hazard characterization

Available epidemiological data are considered inadequate to serve as a basis for the assessment of non-neoplastic effects and carcinogenicity in humans, as a consequence of inconsistencies in findings among the available studies, lack of quantitative monitoring data for the populations studied and/or concomitant exposure to other substances.

In experimental animals, phenol is moderately acutely toxic by ingestion. Owing to its rapid absorption through skin, LD_{50}s for the dermal route are similar to those for oral exposure. Following oral or dermal exposure to acutely toxic doses of phenol, necrosis of the skin or mucous membranes of the throat, neuromuscular tremors and convulsions and histopathological effects on the kidney, liver, spleen and thymus have been observed. Cardiac arrhythmias have also been reported. Phenol is a dermal, ocular and respiratory irritant, but not a skin sensitizer.

Observed effects in limited available studies of repeated-dose toxicity following short-term and subchronic exposure by ingestion have been restricted principally to decreases in body weight gain (often associated with decreases in water consumption), histopathological effects on the kidney, liver and thymus, and myocardial necrosis. Hematological effects, suppressed immune response and neurological manifestations,

TABLE 8 Estimated daily intake of phenol from use of selected consumer products

Product (phenol content)	Estimated intake of phenol	Assumptions
Medicated lip ointment (0.5% w/w, or 5 mg phenol/g ointment) — for treatment of severely dry, cracked lips, cold sores, fever blisters, sunburned lips	4 µg/kg-bw per day	- applied 6 times each day (no data on lip ointment, assume upper end of range of frequency of use reported for lipstick; ECETOC, 1994) - quantity per application 0.01 g (typical quantity per application reported for lipstick; ECETOC, 1994) - 100% absorption (product left on, amount not absorbed likely ingested, phenol readily absorbed dermally, product contains oily vehicle, which would facilitate absorption) - body weight 70.7 kg (average for adults from EHD, 1997)
Ointment A (0.5% w/w, or 5×10^{-3} mg phenol/mg ointment) — for treatment of burns, scalds, sunburn, minor skin irritation	81 µg/kg-bw per day	- applied twice daily to half the surface of forearm (surface area 0.057 m^2, or 570 cm^2) (U.S. EPA, 1996) - typical quantity per application 1 mg/cm^2 (value reported by ECETOC, 1994, for "general-purpose cream") - 100% absorption (product left on, phenol readily absorbed dermally, product contains oily vehicle, which would facilitate absorption; however, estimate is conservative, in that it does not consider that some of ointment would be absorbed onto wound dressing) - body weight 70.7 kg (average for adults from EHD, 1997)
Ointment B (0.18% w/w, or 1.8×10^{-3} mg phenol/mg ointment) — for treatment of cuts, scrapes, minor burns, insect bites, chafing, chapping	29 µg/kg-bw per day	- applied twice daily to half the surface of forearm (surface area 0.057 m^2, or 570 cm^2) (U.S. EPA, 1996) - typical quantity per application 1 mg/cm^2 (value reported by ECETOC, 1994, for "general-purpose cream") - 100% absorption (product left on, phenol readily absorbed dermally, product contains oily vehicle, which would facilitate absorption; however, estimate is conservative, in that it does not consider that some of ointment would be absorbed onto wound dressing) - body weight 70.7 kg (average for adults from EHD, 1997)
Throat lozenges (1.32% w/w) — for relief of sore throat pain and minor mouth irritations	3400 µg/kg-bw per day	- 8 lozenges per day (the maximum recommended on package directions) - 2.25 g/lozenge - 100% absorption - body weight 70.7 kg (average for adults from EHD, 1997)
Sore throat spray (1.4% w/v) — for relief of throat and mouth soreness	3300 µg/kg-bw per day	- 3 sprays per application, 8 applications/day (the maximum recommended on package directions); average weight of the resultant 24 sprays is 16.76 g, based on 5 replicates (relative standard deviation 3.3%) - specific gravity 1 g/mL - assume none exhaled, and 100% absorption - body weight 70.7 kg (average for adults from EHD, 1997)

such as inhibited pupil response, biochemical changes in the brain and behavioural effects, have also been reported.

Although phenol is primarily negative in bacterial tests for mutagenicity, it induces gene mutations and structural chromosomal aberrations in mammalian cells *in vitro*. While results of available studies are mixed, in investigations of optimum design, phenol has induced micronuclei in the bone marrow of mice exposed *in vivo*. On the basis of available data, therefore, it is considered to be a weak *in vivo* clastogen.

Available data, although limited, support the likelihood that phenol is, at most, weakly carcinogenic, based on the lack of clear evidence of carcinogenicity in rats and mice in an early study after administration of relatively high doses in drinking water and the additional observation that it is a weak *in vivo* clastogen and only weakly or not carcinogenic, although a moderate promoter, at non-toxic doses in mouse skin.

Available data are inadequate to serve as a basis for assessment of reproductive effects or of developmental effects following inhalation exposure. In well-conducted studies of the developmental effects of ingested phenol in rats and mice, at maternally toxic doses, there have been reduced litter sizes, increased prenatal loss and perinatal mortality, decreases in average fetal weight per litter, hindlimb paralysis and short or kinky tails. There is some evidence of developmental effects in the absence of maternal toxicity, restricted to a decrease in fetal weights in rats (Jones-Price *et al.*, 1983a).

3.3.3 Dose–response analysis

There are no adequate data with which to characterize dose–response for the potential carcinogenicity of phenol, based on either epidemiological studies or bioassays in experimental animals.

Information on dose–response for non-neoplastic effects of chronic exposure of humans to phenol is limited. Immune response was suppressed and there were effects on hematological parameters in a small number of office workers exposed for six months to a mixture including phenol (approximately 1.3 mg/m^3 air) (Baj *et al.*, 1994). Increases in serum hepatic enzymes and effects on hematological parameters were also observed in a cross-sectional study of a small number of workers in an oil refinery exposed to a time-weighted average concentration of 21 mg/m^3 (Shamy *et al.*, 1994). Primarily as a result of the paucity of data and/or concomitant exposure to other substances, these data are considered inadequate for characterization of dose–response. The remainder of this discussion, therefore, addresses principally information acquired in experimental studies in animal species.

Available data concerning effects following inhalation are considered inadequate to provide meaningful characterization of concentration–response.[7] There are also no recent studies in which a wide range of effects (e.g., clinical, hematological, histopathological effects in a broad range of tissues) has been examined following short-term or subchronic ingestion of phenol.

In contrast, developmental toxicity following oral exposure has been well investigated, and information on dose–response relationships is most extensive in this area, although available data are inadequate to assess reproductive toxicity.[8] However, owing to the lack of observation of effects at several dose levels in studies of developmental effects, development of meaningful benchmark doses is precluded.

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[7] The two-week inhalation toxicity and two-week recovery study of phenol in rats conducted for the Chemical Manufacturers Association (Section 2.4.3.2) was not fully reported at the time of completion of the health risk assessment and was therefore not considered in the hazard evaluation and dose–response analyses for effects on human health.

[8] It is noted that a two-generation oral (drinking water) reproductive study of phenol in rats was being conducted by the Chemical Manufacturers Association at the time of this assessment.

Data on toxic effects (other than tumour promotion) following repeated dermal exposure to phenol have not been identified.

The only investigation in which there were developmental effects in the absence of (well-examined) maternal toxicity was the significant reduction of average fetal body weight in rats at 120 mg/kg-bw per day (water by gavage); the NOAEL was 60 mg/kg-bw per day (Jones-Price *et al.*, 1983a). Although significantly reduced litter sizes were reported in a more recent study in which rats were exposed to a lower dose (40 mg/kg-bw per day in water by gavage) on days 6–19 of gestation (Narotsky and Kavlock, 1995), maternal toxicity (significant reduction in body weight gain and altered respiration; food consumption not reported), not observed at such low doses by others, was also evident.

In a limited early study of repeated-dose toxicity, "slight" changes in the liver and "slight to moderate" kidney damage were observed in rats receiving 50 (liver) or 50 or 100 (kidney) mg phenol/kg-bw per day (gavage in a 1% water solution) over six months (Adams, 1944); incidence or statistical significance of these lesions was not reported. These results contrast with those of early subchronic and chronic studies of the National Toxicology Program (NCI, 1980), in which the only adverse effects observed were decreases in body weight gain in rats and mice at much higher doses, although histopathological effects in a range of tissues were examined (NOAELs in subchronic study greater than 100 and 200 mg/kg-bw per day in mice and rats, respectively; in chronic study, LO(A)EL in mice = 356 mg/kg-bw per day; NOAEL in rats = 280 mg/kg-bw per day). This difference is unlikely to be attributable to variation in the mode of administration (drinking water for the latter versus bolus doses by gavage in water solution for the former), although it is consistent with data on the profile of metabolites observed in a recent toxicokinetic study following exposure via gavage in water or ingestion in drinking water (Hiser *et al.*, 1994), since bolus doses are above the level of saturable metabolism. The more limited conjugation capacity of immature rats may also explain some observed variations in effect levels for animals of different ages. In a more recent short-term study with histopathological examination of the liver, kidney, spleen, thymus and adrenals of younger rats (exposure initiated at 70 days), non-significant (based on only eight rats per group) increases in histopathological changes in the kidney were observed at the highest dose administered by gavage in water at which any animals survived (LOEL = 40 mg/kg-bw per day; NOAEL = 12 mg/kg-bw per day) (Berman *et al.*, 1995).

Also, although not well examined, phenol has induced immunological effects at doses within the range mentioned above, in recent short-term studies. Berman *et al.* (1995) reported non-significant increases in thymic necrosis in small groups of rats (n = 8) at 12 and 40 mg phenol/kg-bw per day received by gavage in water for 14 days (NOAEL unclear; Benchmark $Dose_{05}$ = 9.3 mg/kg-bw per day; lower 95% confidence limit = 3.1 mg/kg-bw per day).[9] In a recent well-documented four-week study by Hsieh *et al.* (1992), immune response was suppressed at doses of 6.2 mg/kg-bw per day and higher; although there were no gross lesions or weight changes observed in the spleen or thymus at doses up to 33.6 mg/kg-bw per day, histopathological examination was not conducted.[10] It is also of interest that suppression of immune response was observed in a study of a small number of workers exposed for six months to phenol as well as formaldehyde and possibly other chemicals emitted from Ksylamit® (used for protection of felt plates inside the office building in which they worked) (Baj *et al.*, 1994).

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[9] The incidence of thymic necrosis at 0, 4, 12 and 40 mg/kg-bw per day was 0/8, 0/8, 1/8 and 2/8, respectively. Because of the small group sizes, the NOAEL is unclear, but it might be considered as 12 mg/kg-bw per day.

[10] Interestingly, in another study conducted by the same authors in the same strain of mice (Hsieh *et al.*, 1988), benzene, for which phenol is a principal metabolite, suppressed immune response at higher concentrations (40 mg/kg-bw per day).

Also, although not well examined, neurobiochemical and behavioural effects have been observed at doses within the same range, with the NOEL for behavioural effects being 12 mg/kg-bw per day in a two-week study in rats (Moser *et al.*, 1995). The toxicological significance of the inconsistent hematological (Sudakova and Nosova, 1981; Hsieh *et al.*, 1992) and neurobiochemical effects (Hsieh *et al.*, 1992) observed at lower doses is unknown.

A Tolerable Intake (TI) has been derived, therefore, based on division of an effect level by uncertainty factors recognizing that dose–response has been best characterized for developmental toxicity, but taking into account limited available data on other endpoints. It is recommended, however, that this value be reconsidered upon completion of more extensive ongoing repeated-dose inhalation toxicity and multigeneration reproductive studies. NO(A)ELs for developmental effects and histopathological effects in the kidney are greater than or equal to 12 mg/kg-bw per day. Indeed, this value might be considered conservative in view of the fact that only non-significant increases in histopathological effects of the kidney were observed at the next (three-fold) higher dose, although group sizes in the critical study were small (n = 8). Although there have been reports of immunological effects at similar or lower concentrations (Hsieh *et al.*, 1992), it was considered premature to use this unconfirmed evidence as a basis for development of a measure of dose–response at this time due to the unknown influence of stress in inducing the observed responses and the questionable clinical significance of the observed effects. On this basis, a TI has been derived as follows:

$$\text{TI} = \frac{\text{12 mg/kg-bw per day}}{100}$$

$$= \text{0.12 mg/kg-bw per day, or}$$
$$\text{120 µg/kg-bw per day}$$

where:

- 12 mg/kg-bw per day is the NOAEL for histopathological effects in the kidney in available studies of adequate, although suboptimal (i.e., small group sizes), design (Berman *et al.*, 1995); this value is less than identified NOELs for developmental toxicity for which dose–response is relatively well characterized and less than NOELs in longer-term studies of repeated-dose toxicity,
- 100 is the uncertainty factor (×10 for intraspecies variation; ×10 for interspecies variation). Available data were considered inadequate to replace components of the default uncertainty factors for inter- and intraspecies variations with data-derived values; however, if hydroquinone is the active metabolite (although this has not been established with certainty), humans are likely to be less sensitive than rats in view of the approximately 20-fold less production in rats compared with humans.

The TI is also considered to be conservative in view of the fact that the critical effect level is based on a study in which administration was by gavage of a bolus dose at a dose within the saturable range of metabolism. Moreover, only non-significant increases in histopathological effects in the kidney were observed at the LOEL, which was approximately three-fold higher than the NOAEL on which the TI is based. In view of the degree of conservatism of the TI, no additional component to address the lack of an adequate study of reproductive effects has been incorporated into the uncertainty factor. It should also be noted that a TI based on the LOEL in the critical study divided by an appropriate uncertainty factor (such as 500) would be similar to that presented here.

In view of the limitations of available data concerning effects following inhalation, which are considered inadequate to provide meaningful characterization of concentration–response, and recognition that a two-week inhalation toxicity and recovery study is under way, a Tolerable Concentration (TC) for inhalation has not been developed. It is noted, though, that results of a recent toxicokinetic study indicate that the profile of metabolites was similar following exposure to

similar doses (on a mg/kg-bw basis) administered either by inhalation or orally as a bolus dose in water for one or eight days (Hiser *et al.*, 1994).

3.3.4 Human health risk characterization

The total daily intake of phenol from all environmental media for the various age groups in the Canadian general population is estimated to range from 0.06 to 0.71 µg/kg-bw per day. These estimated intakes are 169- to 2000-fold less than the TI derived above. Based on the maximum measured concentration of phenol in the air near point sources in Canada. total exposure is estimated to range from 12 to 34 µg/kg-bw per day for these high-exposure subgroups located in the vicinity of point sources. These estimated intakes are 3.5- to 10-fold lower than the TI derived above. Based on the maximum estimated 24-hour concentration of phenol from air dispersion modelling of the largest source in Canada, worst-case estimates of intake range from 29 to 86 µg/kg-bw per day, which approaches the TI of 120 µg/kg-bw per day. However, such high concentrations of phenol were predicted to occur only rarely (0.1% of the time), and only close to the stack and on the roof of the facility (predicted concentrations decreased substantially as distance from the stack increased). Therefore, populations living nearby are unlikely to be exposed to concentrations as high as those predicted immediately next to the facility, and it is considered unlikely that the actual exposures in this situation will approach the TI.

3.3.5 Uncertainties and degree of confidence in human health risk characterization

There is a high degree of uncertainty inherent in the estimates of the intake of phenol in food, the likely principal medium of exposure, due to the relatively high detection limits in the Canadian survey used to estimate exposure from this medium. Perhaps as a consequence of this analytical insensitivity, phenol was not detected in the majority of food composites analysed in this survey. This degree of uncertainty has been characterized quantitatively by calculating intakes on the basis of the assumption of zero or detection limit for foods in which phenol was not detected. The resultant range of intakes is as much as an order of magnitude for some age classes.

There is also a high degree of uncertainty in the estimates of intake in ambient air. Because phenol was not accurately quantified in the available Canadian surveys of ambient air removed from point sources, the estimated intakes via air for the general population were based on data collected in 1974 from a single U.S. urban/suburban site. There are more recent data on concentrations of phenol in the vicinity of point sources in Canada, but the available studies are very limited in scope. All were conducted over very short time periods, in most instances at very few locations and often with little or no indication of the location of monitoring in relation to local human populations. In addition, a number were conducted downwind of sources. It is also clear from these surveys that levels near point sources vary tremendously, depending on the source type and location. In addition, the estimates near the largest source were based on air dispersion modelling (in which chemical transformations are treated to a limited extent and phenol is assumed to be in the aerosol state) and a maximal emission rate, rather than monitoring data. For these reasons, there is considerable uncertainty as to the actual exposures experienced by populations near point sources in Canada, and the upper-bound worst-case estimates of intake were conservative — i.e., based on assumptions that would yield maximal values.

However, there is a fair degree of certainty that drinking water and soil contribute only negligible amounts to the total exposure to phenol. Based on data from extensive surveys of drinking water and more limited surveys of soils from non-contaminated areas, even the upper ends of the ranges of estimated intakes via these media, calculated by assuming that phenol was present at the detection limit in samples in which it was not detected (the vast majority), are orders of magnitude lower than those in food or ambient air.

The overall degree of confidence in the population exposure estimates is, therefore, low, owing principally to the lack of current, representative monitoring data for food, the likely

principal medium of exposure of the general population in Canada, and for ambient air in the general environment and in the vicinity of a range of point sources.

The degree of confidence in the database on toxicity that serves as the basis for development of the TI is low to moderate. The epidemiological data in humans are inadequate. There are also no recent repeated-dose toxicity studies in animals in which a range of endpoints has been well characterized by current standards,[11] with the exception of developmental toxicity, investigated in well-conducted studies in rats and mice. Moreover, some additional uncertainty is introduced by reports, following short-term exposure to phenol at doses less than the NO(A)EL on which the TI is based, of suppression of immune response and/or hematological changes in mice or guinea pigs, and of neurobiochemical effects in rats (although there are only single reports, and the effects are of unknown toxicological significance). In addition, available data on reproductive effects are quite limited.[12]

3.4 Conclusions

CEPA 11(a): Based on available data, it has been concluded that phenol is not entering the environment in a quantity or concentration or under conditions having or that may have an immediate or long-term harmful effect on the environment. Therefore, phenol is not considered to be "toxic" as defined in Paragraph 11(a) of CEPA.

CEPA 11(b): Based on available data, it has been concluded that phenol is not entering the environment in a quantity or concentration or under conditions constituting or that may constitute a danger to the environment on which human life depends. Therefore, phenol is not considered to be "toxic" as defined in Paragraph 11(b) CEPA.

CEPA 11(c): Based on available data, "it has been concluded that phenol is not entering the environment in a quantity or concentration or under conditions constituting or that may constitute a danger in Canada to human life or health." Therefore, phenol is not considered to be "toxic" as defined in Paragraph 11(c) of CEPA.

Overall conclusion: Based on critical assessment of relevant information, phenol is not considered to be "toxic" as defined in Section 11 of CEPA.

3.5 Considerations for follow-up (further action)

Since phenol is not considered to be "toxic" as defined in Section 11 of CEPA, investigation of options to reduce exposure under CEPA is not considered a priority at this time. However, this is based upon current use patterns; future releases of this compound should continue to be monitored to ensure that exposure does not increase to any significant extent.

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[11] It is noted that a two-week inhalation and two-week recovery study and a two-generation oral (drinking water) reproductive study have been conducted for the Chemical Manufacturers Association but were not yet fully reported at the time of completion of the health assessment. When available, the results of these studies should be evaluated with respect to their implications for designation of "toxic," but in view of other priorities for assessment under CEPA.

[12] It is noted that a two-generation oral (drinking water) reproductive study of phenol in rats was being conducted by the Chemical Manufacturers Association at the time of this assessment.

Available data indicate that releases of phenol into the environment in Canada are greatest for the pulp, paper and wood, mineral (non-metallic), chemical, steel and metal, and petroleum refining and products sectors (Table 2).

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APPENDIX A SEARCH STRATEGIES EMPLOYED FOR IDENTIFICATION OF RELEVANT DATA

Environmental assessment

Data relevant to the assessment of whether phenol is "toxic" to the environment under CEPA were identified from existing review documents and searches of commercial and government databases conducted between January and April 1996. The major databases that were searched include the following: Aqualine (1985–1996), ASFA (Aquatic Sciences and Fisheries Abstracts, Cambridge Scientific Abstracts; 1990–1996), BIOSIS (Biosciences Information Services; 1990–1996), CAB (Commonwealth Agriculture Bureaux; 1990–1996), CESARS (Chemical Evaluation Search and Retrieval System, Ontario Ministry of the Environment and Michigan Department of Natural Resources; 1996), CHRIS (Chemical Hazard Release Information System; 1996), Current Contents (Institute for Scientific Information; 1990–1996), ELIAS (Environmental Library Integrated Automated System, Environment Canada library; January 1996), Enviroline (R.R. Bowker Publishing Co.; 1990–1996), Environmental Abstracts (1975 – February 1996), Environmental Bibliography (Environmental Studies Institute, International Academy at Santa Barbara; 1990–1996), GEOREF (Geo Reference Information System, American Geological Institute; 1990–1996), HSDB (Hazardous Substances Data Bank, U.S. National Library of Medicine; 1996), IRPTC (International Register of Potentially Toxic Chemicals; April 1996), Life Sciences (Cambridge Scientific Abstracts; 1990–1996), NPRI (National Pollutant Release Inventory; 1995), NTIS (National Technical Information Service, U.S. Department of Commerce; 1990–1996), Pollution Abstracts (Cambridge Scientific Abstracts, U.S. National Library of Medicine; 1990–1996), POLTOX (Cambridge Scientific Abstracts, U.S. National Library of Medicine; 1990–1995), RTECS (Registry of Toxic Effects of Chemical Substances, U.S. National Institute for Occupational Safety and Health; 1996), Toxline (U.S. National Library of Medicine; 1990–1996), TRI93 (Toxic Chemical Release Inventory, U.S. Environmental Protection Agency, Office of Toxic Substances; 1993), USEPA-ASTER (Assessment Tools for the Evaluation of Risk, U.S. Environmental Protection Agency; up to December 21, 1994), WASTEINFO (Waste Management Information Bureau of the American Energy Agency; 1973 – September 1995), Water Resources Abstracts (U.S. Geological Survey, U.S. Department of the Interior; 1990–1996). Additional data sources are listed in Appendix F of the environmental supporting documentation (Environment Canada, 1998a). A survey of Canadian industry was carried out under authority of Section 16 of CEPA (Environment Canada, 1997d). Companies were required to provide information on uses, releases, environmental concentrations, effects or other data that were available to them for phenol if they met the trigger quantity of 500 kg of phenol per year. Additional relevant information was obtained on a voluntary basis from industry, including several steel and metal products mills. Reveal Alert was used to maintain an ongoing record of the current scientific literature pertaining to the potential environmental effects of phenol. Data obtained after May 31, 1998, were not considered in this assessment unless they were critical data received during the 60-day public review of the report (May 1 to June 29, 1999).

Health assessment

Evaluations of other agencies, such as the International Programme on Chemical Safety (IPCS, 1994a), the Commission of the European Communities (Hansen, 1993) and the Agency for Toxic Substances and Disease Registry (ATSDR, 1989), were consulted to identify relevant data.

Additional relevant data were identified through searches, conducted in January 1994, on the following databases: CCRIS (Chemical Carcinogenesis Research Information System, U.S. National Cancer Institute), DART (Developmental and Reproductive Toxicology, U.S. National Library of Medicine), EMIC (Environmental Mutagen Information Center database, Oak Ridge National Laboratory) and EMICBACK (backfile of EMIC), ETICBACK (backfile of Environmental Teratology Information Center database, U.S. Environmental Protection Agency and U.S. National Institute of Environmental Health Sciences), GENE-TOX (Genetic Toxicology, U.S. Environmental Protection Agency), HSDB (Hazardous Substances Data Bank, U.S. National Library of Medicine), IRIS (Integrated Risk Information System, U.S. Environmental Protection Agency), RTECS (Registry of Toxic Effects of Chemical Substances, U.S. National Institute for Occupational Safety and Health). Phenol's name, registry number and major synonyms were searched in the Toxline database (U.S. National Library of Medicine; 1988 to present). In view of the relatively limited database available for assessment of the toxicity of phenol, additional attempts were made to acquire copies of unpublished studies through contact with representatives of industry and international agencies and library searches of authors' names to identify subsequently published work. In response to these requests, a copy of the Dow Chemical Company (1944) unpublished manuscript was kindly provided by the Chemical Manufacturers Association. The Chemical Manufacturers Association also provided preliminary reports of ongoing industry-sponsored inhalation and reproductive toxicology studies.

To identify data relevant to the estimation of exposure of the general population to phenol, literature searches were conducted in January 1994 using the strategy of searching by the name and major synonyms in the following databases: AQUAREF (Inland Waters Directorate, Environment Canada), CISTIMON (Canadian Institute for Scientific and Technical Information list of monographs, National Research Council of Canada), ELIAS (Environmental Library Integrated Automated System, Environment Canada library), EMBASE (on-line version of Excerpta Medica), Enviroline (R.R. Bowker Publishing Co.), Environmental Bibliography (Environmental Studies Institute, International Academy at Santa Barbara), Medline (U.S. National Library of Medicine), Microlog (Canadian Research Index, Government Publications, Micromedia Ltd.), Pollution Abstracts (Cambridge Scientific Abstracts, U.S. National Library of Medicine). Numerous provincial and federal government officials and representatives of various industrial sectors were contacted between February and August 1996 for data relevant to exposure and/or effects. Only data acquired prior to September 1997 were considered in the determination of whether phenol is "toxic" to human health.